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SYMPOSIUM ON DIRECT TRACER MEASUREMENT OF THE REAERATION CAPACITY OF STREAM AND ESTUARIES, 7-8 JULY 1970

Ernest C. Tsivoglou, et al

Georgia Institute of Technology

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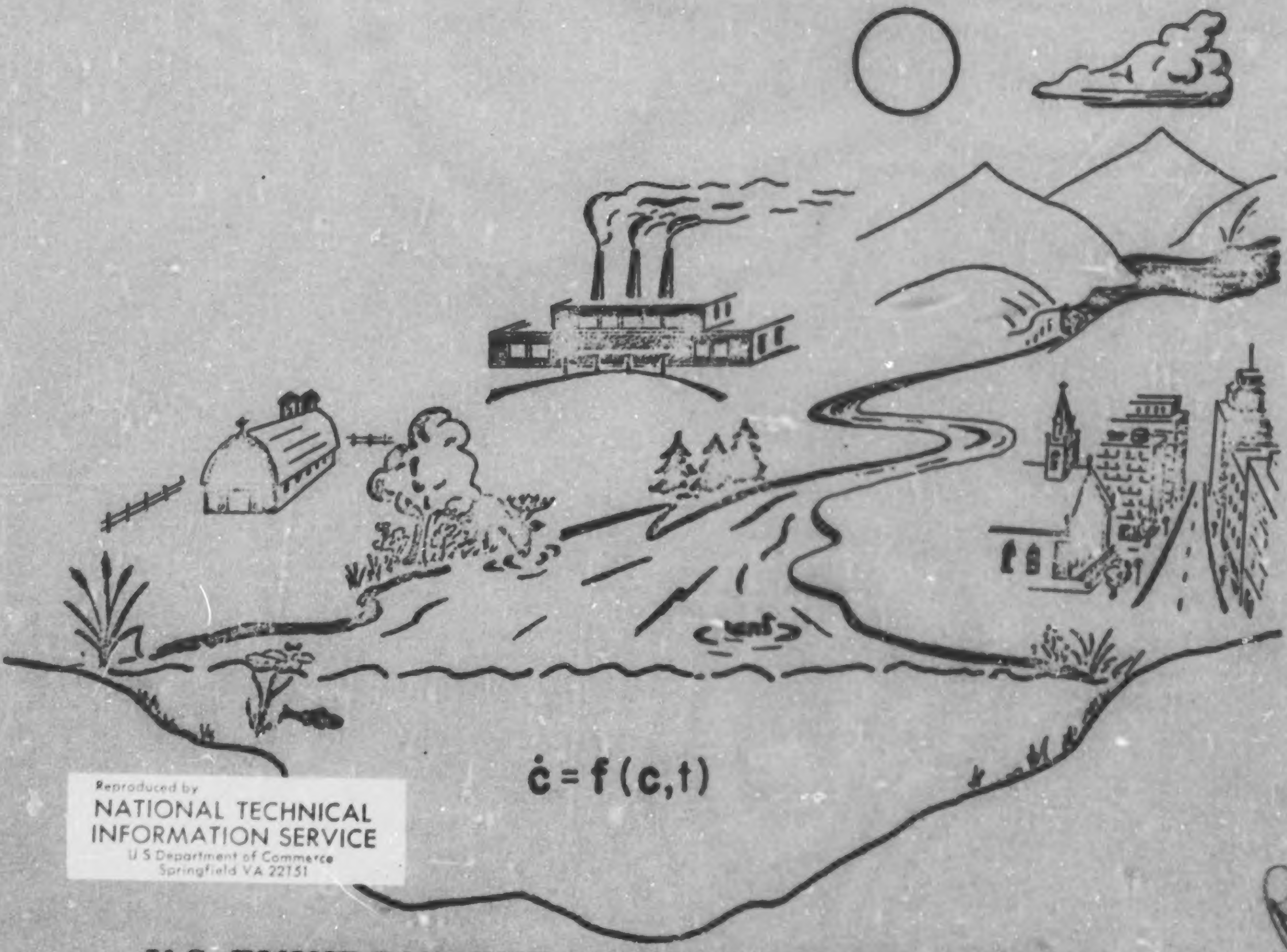

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27 *Abstract*

A symposium on direct measurement of the reaeration capacity of streams and estuaries was conducted in July 1970, for the purpose of making immediately available the results of current research on this subject at the Georgia Institute of Technology. The symposium was designed to make public for the use of other engineers and scientists all of the available information on the subject at that time.

The papers presented provide an outline of the fundamentals of gas transfer in turbulent systems, the theory and application of radiotracers for measuring gas transfer in natural waters, and the associated field and laboratory procedures. Other papers provide tracer-observed values of the reaeration capacity of several streams, and comparisons with computed values obtained from well-known predictive models. A new theory regarding the relationship between the reaeration capacity and the hydraulic properties of natural streams is presented, together with early supporting observed results. The effects of pollutants on the reaeration capacity, and some observed results, are discussed in another paper. Invited papers provide the initial results of tracer measurement of the reaeration capacity of a small estuary, as well as the oxygen balance for an inland stream using the tracer-observed reaeration capacity (by Georgia Tech) together with DO and BOD data obtained independently (by EPA). (Tsivoglou-Georgia Tech)

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PROCEEDINGS
# of a
SYMPOSIUM
on

DIRECT TRACER MEASUREMENT
OF THE
REAERATION CAPACITY OF STREAMS AND ESTUARIES

July 7-8, 1970

COSPONSORS

Environmental Protection Agency

and

The Georgia Institute of Technology
School of Civil Engineering

SYMPOSIUM ARRANGEMENT

Ernest C. Tsivoglou, Principal Investigator, GIT

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January, 1972

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ABSTRACT

A symposium on direct measurement of the reaeration capacity of streams and estuaries was conducted in July, 1970, under the joint sponsorship of the Georgia Institute of Technology, School of Civil Engineering, and the Environmental Protection Agency, for the purpose of making immediately available the results of current research on this subject at Georgia Tech. The symposium was designed to make public for the use of other engineers and scientists all of the available information on the subject at that time.

The papers presented and included here provide an outline of the fundamentals of gas transfer in turbulent systems, the theory and application of radiotracers for measuring gas transfer in natural waters, and the associated field and laboratory procedures. Other papers provide tracer-observed values of the reaeration capacity of several streams, and comparisons with computed values obtained from well-known predictive models. A new theory regarding the relationship between the reaeration capacity and the hydraulic properties of natural streams is presented, together with early supporting observed results. The effects of pollutants on the reaeration capacity, and some observed results, are discussed in another paper. Invited papers provide the initial results of tracer measurement of the reaeration capacity of a small estuary, as well as the oxygen balance for an inland stream using the tracer-observed reaeration capacity (by Georgia Tech) together with DO and BOD data obtained independently (by EPA).

These Proceedings thus reflect the state-of-the-art of measuring and predicting the reaeration capacity of natural streams as of July, 1970.

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FOREWORD

This special symposium on tracer measurement of the reaeration capacity of streams and estuaries was arranged and conducted in the interests of immediate technology transfer, as an outgrowth of EPA Research Grant 16050 EDT, "Characterization of Stream Reaeration Capacity." During the course of that research it was decided, together with the Project Officer, Dr. Walter M. Sanders,III, that the results already obtained and the techniques in use had sufficient scientific and technological significance to warrant making this information available without delay to engineers and scientists in the field. Accordingly, the symposium was arranged. The material presented here represents all of the information available as of the date of the symposium. Research under Project 16050 EDT continued thereafter, and the complete results of that research are presented in the subsequent report for that Project.

Ernest C. Tsivoglou,
Principal Investigator

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SECTION I

CONCLUSIONS

The tracer method for measuring gas transfer in natural streams and estuaries makes it possible for the first time to obtain independent field measurements of stream reaeration capacity. The technique is fully developed and effective for application in nontidal streams. Initial application in a small estuary has indicated some difficulty associated with stratification of fresh and salt water flow, and for such application considerable care must be exercised in locating the most useful tracer release depth, as the tracer will faithfully reflect the gas transfer that takes place in those volume elements of water that are actually dosed.

For highly turbulent streams, it has been shown that much of the real action of gas transfer may take place in quite short reaches and times, and for such streams none of the available models for predicting reaeration on the basis of hydraulic properties provide predictions that approach the observed reaeration coefficients. For less turbulent streams of smaller slope, several of the available predictive models yield predicted values more nearly in accord with observed reaeration coefficients.

A new model relating reaeration capacity directly to energy dissipation has been proposed, based on the results of current research at Georgia Tech. In this model, energy dissipation is evaluated in terms of the change in water surface elevation between two stream locations and the time of flow. This model has provided predictions in good agreement with observed results over the entire range of observation, and current research at Georgia Tech is directed toward further improvement and refinement of this basic model relating reaeration capacity to measurable stream hydraulic properties.

Studies of the effects of pollutants on the reaeration capacity have also been conducted, both with specific pollutants in distilled water and with natural stream samples that contain mixed pollutants. The initial results of this research indicate that detergents (LAS) decidedly reduce the reaeration capacity, depending upon the concentration and the degree of turbulent mixing, and have demonstrated that the addition of partially treated domestic sewage to a stream can significantly reduce the natural reaeration capacity of the stream.

SECTION II

RECOMMENDATIONS

As a result of this state-of-the-art review of available information on measuring the reaeration capacity of natural streams, it is recommended that current research in this field be directed toward developing improved understanding of the basic relationships between energy dissipation and reaeration in natural streams, and, specifically, toward the final development of a fully satisfactory model for predicting stream reaeration capacity in terms of hydraulic properties that can be measured directly and with accuracy.

It is further recommended that additional research be conducted on the important matter of the effects of various pollutants on the reaeration capacity of natural streams. Such research should be directed toward the development of improved understanding of pollutant effects, to the point that such effects can be predicted with accuracy and confidence and that the real environmental damages associated with these effects can properly be evaluated and assigned to the responsible pollution sources.

It is recommended also that a second symposium on natural stream reaeration be conducted at an appropriate time when sufficient new information is available regarding hydraulic models for reaeration capacity, the effects of pollutants on the reaeration rate coefficient and on the oxygen saturation limit, and stream oxygen balances based upon known reaeration capacity.

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SECTION III

PAPERS

Turbulence, Mixing and Gas Transfer

E. C. Tsivoglou

Introduction and Background

Reaeration refers to the ability of a flowing stream, or any other turbulent water system, to obtain oxygen from the limitless resource of the atmosphere. Some of you here are very familiar with this subject and some are not, so that a brief outline of the state of the art and the principles of gas transfer in turbulent streams appear desirable before proceeding to a discussion of the results of our research.

Stream self-purification involves two principal processes, namely: (a) the depletion of dissolved oxygen resources due to bacterial degradation of domestic and industrial organic wastes; and (b) replenishment of the dissolved oxygen resource by absorption of oxygen from the atmosphere. The second process, reaeration, is a direct function of turbulence, but we have no way of measuring turbulence independently.

Other natural processes modify the oxygen balance in a polluted stream or reservoir: the anaerobic decomposition of benthal deposits of settleable organic matter results in a local demand on stream DO resources; if algae are present in large numbers, they will add oxygen to the stream during daylight hours by photosynthesis and will consume DO during the dark hours by respiration; in some streams, prolific growths of attached bacterial slimes have a great influence on the oxygen balance in the flowing water; the situation is often further complicated by the presence of multiple sources of pollution and tributary flows. All of these oxygen-influencing processes occur simultaneously in a polluted stream, to lesser or greater degree, and in a specific case any one of them may dominate the total self-purification process.

Stream self-purification is thus a very complex process in any real situation. Unfortunately, we do not have methods for the independent evaluation of each of the above oxygen-influencing processes, and that is why we have been unable until recently to obtain accurate evaluations of oxygen uptake from the atmosphere—we do not know how to evaluate turbulent mixing, which controls reaeration, and we cannot obtain accurate independent evaluations of some of the other processes such as photosynthesis and bioextraction.

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Indirect Evaluation of Reaeration - Prior to the development of the tracer method for direct and system-independent evaluation of gas transfer, in 1966, all estimates of oxygen income by stream reaeration in real situations had to be made by an indirect oxygen balance procedure. In essence, an attempt is made to evaluate all of the other processes that have influenced the observed stream DO profile, and then a calculation is made of what the reaeration oxygen income must have been in order to produce the observed DO profile. The approach is much the same as that used in estimating the bottom "roughness" of a stream in calculations related to open channel flow - one cannot obtain a system-independent direct measure of roughness, either.

Application of the indirect method of estimating stream reaeration requires that a mathematical model for the observed DO profile be available, and, of course, the development of such a model requires that certain simplifying assumptions be made. The earliest such model was the famous oxygen-sag equation provided by Streeter and Phelps in 1925 (1). This still stands, more than fifty years later, as one of the greatest single contributions in the field, and is still widely used. Its logic is clear, simple and faultless. Of course, as with any such model of natural processes, it is idealized, and the simplifying assumptions that were necessary in order to develop the model also limit its application and effectiveness - and this was recognized by Streeter and Phelps, as well as by others who came later. Nevertheless, the simple oxygen-sag model provides the necessary basis for understanding stream self-purification.

The oxygen-sag model incorporates only the effects of bacterial degradation of the dissolved organic pollution and reaeration, and neglects benthal decomposition, photosynthesis, bottom slimes and such secondary processes. So that it is a little too simplified and idealized for practical application in many of today's pollution problems. For example, if an organic sludge deposit is present, and we ignore it in using the oxygen-sag equation for an indirect estimate of reaeration, we calculate a reaeration oxygen income that is not quite right. In addition to such errors of omission, even the process by which the stream bacteria degrade the dissolved organic pollution may, in some cases, be more complex than envisioned in 1925 by Streeter and Phelps. One cannot ignore, either, the real errors of field and laboratory measurement that creep into any real study of a natural situation.

The net result of such unavoidable errors of assumption, omission and field measurement is that indirectly calculated estimates of reaeration income contain an unknown degree of error, small in some cases and undoubtedly large in others. In point of fact, the reaeration rates calculated by the indirect method contain an error that simply compensates for all of the other errors of assumption, omission and measurement that have been made. As a result, it has not been possible to accept such indirect estimates of reaeration as firm or accurate.

Predictive Models - Because of the above-noted problems of indirectly evaluating stream reaeration, various investigators have attempted over the past 60 years to develop rational mathematical models for the reaeration process itself. Such models generally attempt to explain reaeration in terms of turbulence theory and stream hydraulic properties such as velocity and depth. The first such model was provided in 1911 by Black and Phelps, in a report on the pollution of New York Harbor (2). That model, which attempted to explain reaeration in terms of molecular diffusion, stream depth and a "mixing period", is still in use today (3).

Since 1911, other attempts have been made to explain reaeration in terms of the hydraulic properties that are associated with turbulent mixing. Some of the better known models include those of Streeter and Phelps (1925),(1) O'Connor and Dobbins (1956)(4) and Churchill et al. (1962) (5), all of which consider reaeration (and turbulence) to be directly related to stream velocity and inversely related to stream depth. Other models include that of Krenkel and Orlob (6) who attempted to explain reaeration in terms of longitudinal dispersion, and the Thackston model (7) which incorporates hydraulic slope as an additional factor.

All such mathematical models for stream reaeration are referred to here as predictive models, rather than indirect, as their purpose is to predict reaeration independently in terms of hydraulic factors. In all cases, their development has been hampered and limited because the only means of testing the model has been indirect calculation of the real reaeration income by the questionable oxygen-sag approach. Hence, all of the predictive models must still be regarded as possible but not proved.

The predictive models for reaeration will be discussed in greater detail later. They are regarded as most important, as they represent the necessary direction of development, that is, the explanation of reaeration, and the ability to predict it, in terms of hydraulic properties. Thus, although any or all of the predictive models may prove eventually to be not quite adequate or correct, all of them provide necessary emphasis and insight into the important relationships between reaeration, gas transfer, and turbulent mixing in natural streams.

Purposes of This Research - Recognizing the real need for an independent means of evaluating stream reaeration capacity with accuracy and dependability, in 1964 the Federal Water Pollution Control Administration began studies to develop such a procedure. The result of those studies has been the gaseous tracer method that forms the basis of the studies to be described at this symposium. The tracer method for reaeration was first demonstrated in the field during 1966, in studies of the pollution and self-purification of the Jackson River below Covington, West Virginia. Those field studies demonstrated the techniques and effectiveness of the reaeration tracer procedure, and produced the first direct and independent observations of stream reaeration capacity.

The research reported at this symposium was begun in 1968, and has been sponsored by a grant from the Federal Water Pollution Control Administration to Georgia Tech. It has been conducted through our School of Civil Engineering by the Sanitary Engineering staff and Georgia Tech students. The purposes of the research have been:

(1) To evaluate and define the basic relationships between natural stream reaeration capacity and measurable stream hydraulic properties, by means of direct field tracer and physical studies in local streams.

(2) As a part of the research, to evaluate currently available predictive models for reaeration, in terms of accuracy, range of error, etc.

(3) As needed, to develop modified predictive models, or additional models, for predicting stream reaeration capacity on the basis of measurable hydraulic properties.

(4) To develop a standardized procedure for evaluating the effects of pollutants on stream reaeration capacity, and to apply this technique to evaluate the effects of various pollutants such as detergents, oils, municipal wastes, etc.

As an incidental but not negligible purpose, the research program would also provide useful reaeration data for real pollution problems in the Atlanta and Georgia Tech vicinity.

Fundamentals of Gas Transfer

Reaeration is a purely physical process that involves: (1) entry of oxygen molecules from the atmosphere into the water at the air-water interface; and (2) subsequent distribution of this dissolved oxygen throughout the volume and depth of water. The driving force for reaeration (or for the transfer of any other gas) is simply the partial pressure difference of oxygen between the atmosphere and the water. When the water achieves a partial pressure of dissolved oxygen that is equal to the partial pressure of oxygen in the atmosphere, the water is said to be saturated with oxygen, and there will be no further net oxygen transfer.

The entry of oxygen molecules into the water from the atmosphere, and their subsequent distribution throughout the water volume and depth involve: (1) molecular diffusion and (2) dispersion, or mixing. Diffusion and dispersion are two quite different processes, although they complement each other, and the technical literature is somewhat confusing at this point. For instance, the commonly-used terms eddy diffusion and hydrodiffusion really refer to dispersion or mixing, rather than to the molecular diffusion process so well known in science. It is also important to bear in mind throughout this discussion that the water can obtain additional oxygen only at the air-water interface, or the water surface. Figures 1 through 4 provide a brief outline of the fundamentals of the diffusion and dispersion processes, and of their respective roles in reaeration and gas transfer.

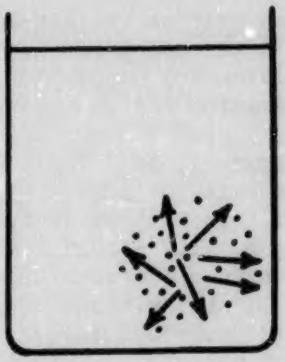
Molecular Diffusion - Referring to Figure 1, if I could place a group of dissolved molecules (such as salt, or a gas) at some point in a beaker of stagnant water, and do this without disturbing the water, the dissolved molecules would: (a) gradually spread out through the water, and (b) eventually achieve a uniform concentration throughout the water in the beaker. They would do this without any movement at all of the water itself, or in totally quiescent water. They would do it because of their own inherent kinetic energy. This is the process referred to as molecular diffusion.

Referring again to Figure 1, all molecules possess inherent kinetic energy associated with their surrounding temperature, and the average kinetic energy is just 3/2 kT, where k is the Boltzmann Constant, and T is the absolute temperature. In terms of mass and velocity, then, molecules of a specific mass will move about with a specific velocity, on the average, according to the model $KE = 1/2\ mv^2$. The dissolved salt or gas molecules therefore move about as shown by the arrows in Figure 1, and this motion is entirely random and takes place in random directions. It is this movement due to inherent kinetic energy that allows the dissolved molecules to spread out and achieve uniform concentration in the beaker of water, by molecular diffusion.

Fick's first law of diffusion places molecular diffusion on a quantitative basis. Referring to Figure 1, J is the net flux of molecules (in $mg/cm^2/sec$) across any plane within the volume of water; dc/dr refers to the concentration gradient across the plane (dc represents the difference in concentration of dissolved material on the two sides of the plane, and dr represents the infinitesimal distance from one side of the plane to the other), and is the driving force for diffusion; D_m is referred to as the coefficient of molecular diffusion, and its magnitude depends upon the molecular characteristics of both the diffusing molecules and the surrounding medium.

In 1905, Albert Einstein developed an equation for evaluation of the molecular diffusion coefficient, based upon his studies of the Brownian motion. Referring to Figure 1, the diffusion coefficient, D_m, is seen to be equal to the product of the universal gas constant, R, and the absolute temperature, T, divided by Avogadro's number, N_o, and a "friction factor", f, related to the ability of the surrounding medium to impede the progress of the diffusing molecule.

A little later, Stokes further defined the friction factor, f, for spherical particles falling freely through water, and showed the friction factor to be directly proportional to the viscosity, η, of the medium and the radius, r, of the falling sphere. Hence, the diffusion coefficient, D_m, is seen to be a function of the absolute temperature, the viscosity of the fluid and the size of the diffusing particle.



$$\overline{KE} = \frac{3}{2} kT$$

Fick's First Law:

$$J = -D_m \frac{dc}{dr}$$

$$(mg/cm^2/sec) = (cm^2/sec) \times \left(\frac{mg/cm^3}{cm}\right)$$

$$D_m = \frac{RT}{N_o f} \qquad \text{(Einstein)}$$

$$f = 6\pi\eta r \qquad \text{(Stokes)}$$

Hence,

$$D_m = f(T, \eta, r)$$

and

$$\frac{dc}{dr} = \text{driving force for diffusion}$$

FIGURE 1

MOLECULAR DIFFUSION

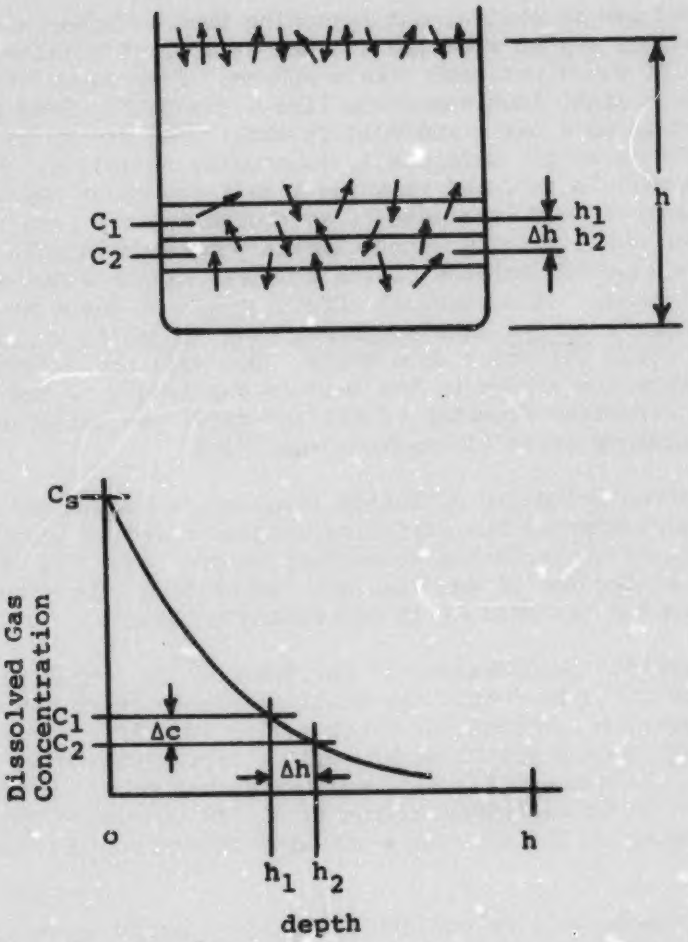
Now a word about gas molecules, and regarding them as spheres. If we could take a single oxygen molecule and set it down on a table, and hold it still, it would not look like a sphere. Presumably, this diatomic molecule might look something like a dumbbell. However, one cannot set it down on a table and hold it still long enough to look at it, because this single molecule is constantly in motion. First, it has what we call "spin", and it spins like a top about an axis; secondly, the axis itself "precesses", as though the top were wobbling, about some other axis; thirdly, the molecule possesses "dipole moments" and "quadrupole moments" related to the movement of the atoms with respect to each other. The combined effect of all of these motions is to make the molecule *behave* like a sphere, even though it wouldn't look like one if it could sit still on a table. Nor will the effective diameter of the operating sphere be the same as the length of the quiet dumbbell. The effective diameter of the spherical gas molecule is of the order of angstrom units (1 angstrom = 10^{-8} cm).

To summarize, then, molecular diffusion takes place because of the inherent kinetic energy of the diffusing molecules and in proportion to the magnitude of the existing concentration gradient; the diffusion coefficient is a function of the absolute temperature, the viscosity of the fluid medium and the size of the diffusing molecules.

Figure 2 illustrates the mechanics of gas transfer in completely quiescent water. The water is completely still, there being no temperature gradients, convection currents, or other motion of volume elements of water. (Although such a system might well be impossible to achieve experimentally, the concept is valid and suitable for our purposes here). Initially, there is no dissolved oxygen at all in the water, so that initially oxygen molecules move only into the water from the overlying atmosphere.

A little later, there will be available dissolved oxygen molecules in the upper water layer near the water surface; they also are in constant movement due to their inherent kinetic energy, and they move in random directions. Some of them escape again to the overlying atmosphere, while others thus diffuse to deeper water layers. However, oxygen molecules are able to enter the topmost water layer from the overlying atmosphere *more easily* than they are able to diffuse downward through the fluid medium. As a result, the dissolved gas molecules accumulate fairly rapidly in the uppermost water layers, and those layers become "saturated."

At any time after the start of the experiment, the *net* rate of entry of gas molecules at the air-water interface is just the rate of entry from above (constant, because the overlying atmosphere has constant oxygen concentration) minus the rate of escape back to the atmosphere (proportional to the dissolved oxygen concentration in the uppermost water layer). Because of the relatively rapid accumulation of gas molecules in the topmost water layer, the *net* rate of entry (or, reaeration) soon becomes very small.



$(C_1 - C_2) = \Delta C$ = very small

$\therefore \left(\frac{\Delta C}{\Delta h}\right)$ = very small

$\therefore J \cong - D_m \left(\frac{\Delta C}{\Delta h}\right)$ = very small

FIGURE 2

GAS TRANSFER IN STAGNANT WATER

As a result, the deeper water layers soon become "starved" for oxygen molecules. Referring to Figure 2, across any infinitesimal distance (depth) Δh, the dissolved oxygen concentration difference, $\Delta c = (c_1 - c_2)$, is infinitesimally small. Hence, at any depth and at any time, the driving force for molecular diffusion, the concentration gradient $(\Delta c/\Delta h)$, is very small. Referring back, then, to Fick's law, diffusion of oxygen molecules downward is very slow, and reaeration of truly stagnant water is a very slow process that requires days or weeks before the bottom layers of water approach DO saturation. The whole process is slow because of the blocking action of molecular diffusion.

Turbulent Mixing - Consider now the same beaker of water, but no longer quiescent. Instead, the water is being mixed by some external force (perhaps the beaker is sitting on a vibrating platform). We will be concerned now primarily with volume elements of water, rather than with molecules of oxygen. We define a volume element to be infinitesimally small in the calculus sense, but large enough to contain a very large number of molecules.

Referring to Figure 3, at the start of our experiment the water contains no dissolved oxygen. Volume element No.1 moves up to the water surface from below and remains there for a definite, if very small, period of time. Because it contained no dissolved oxygen, the net rate of entry of gas molecules from the overlying atmosphere is very large - at a maximum - and the volume element gains a relatively large amount of dissolved oxygen before it leaves the surface to move downward to a deeper location. In its downward path it encounters a second volume element of water, No.2, that has never been at the water surface and so contains very little or no dissolved oxygen. Thus, the one volume element contains quite a large amount of dissolved oxygen compared to the other, and at the interface between them there is a large concentration difference, $\Delta c = (c_1 - c_2)$. Hence for that moment across that interface, the driving force for molecular diffusion, $(\Delta c/\Delta r)$, is relatively large, and the transfer of dissolved gas molecules from the one volume element to the other is relatively rapid.

If we now multiply this example by all of the volume elements of water in the beaker, it is clear that mixing greatly speeds the reaeration process. The water surface is constantly replaced by volume elements from below, and hence the blocking action of molecular diffusion is no longer present. The lower water depths are no more starved for dissolved oxygen than the upper locations. The average concentration of dissolved oxygen is at any time the same at all depths and all locations, including the surface, in a homogeneously mixed system, and, hence, the net rate of entry of gas molecules at the water surface remains relatively large until the whole volume of water approaches the DO saturation concentration. Note also that the dissolved oxgyen concentration gradient, $(\Delta c/\Delta r)$, does not now occur in any preferred direction, such as downward. Instead, there is an average concentration gradient throughout the whole volume of water, and it is multidirectional.

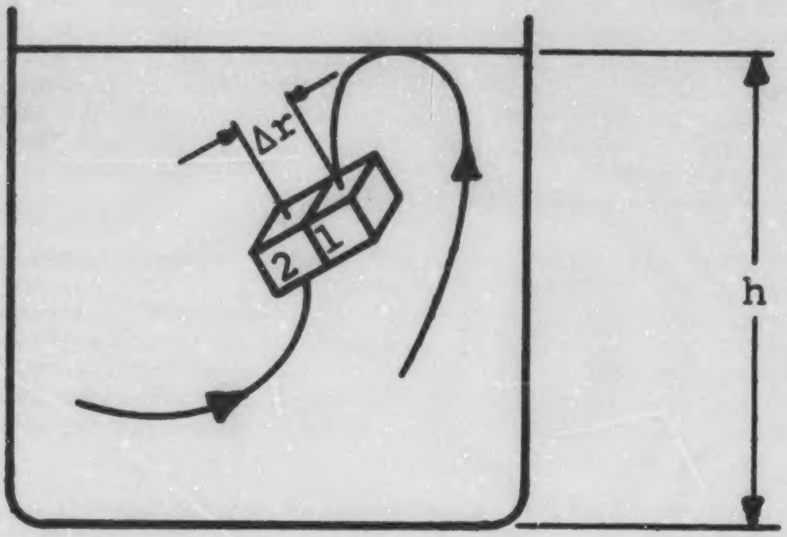
It is also clear that the faster the water is mixed, and the surface replaced, the faster will be the reaeration process. Instead of days or longer, the water can be saturated with dissolved oxygen in minutes at high rates of mix. Thus, molecular diffusion keeps up with mixing in the turbulent system, instead of blocking reaeration. It is also important to note that in the mixed system the depth of water has nothing to do with the rate of reaeration except insofar as the depth-to-volume ratio influences the physical rate of water surface replacement.

So far as reaeration is concerned, then, the term "turbulence" has a special meaning relating strictly to the rate of water surface replacement and to the dispersion of volume elements of water. Turbulent mixing of the water and consequent dispersion of the dissolved gas molecules takes place due to the application of external forces, such as the platform vibration, or a mechanical stirrer, etc. It enhances molecular diffusion and reaeration as outlined above.

Misconceptions - The foregoing outline of the fundamental mechanisms of gas transfer in turbulent water systems indicates that certain widely held concepts of gas transfer are not, in fact, correct representations of the physical facts. In the first place, in a well-mixed system, or in a turbulent natural stream, the surface water layer is not saturated with dissolved oxygen - constant surface replacement precludes this. Also, as indicated above, there is no preferred direction of oxygen transfer, such as downward, and the physical depth of a watercourse influences reaeration only to the extent that it influences the rate of water surface replacement in the hydraulic sense.

In particular, in a homogeneously mixed system no stagnant surface water "film" can exist for any finite period of time, and, hence, even though it may be an adequate mathematical convenience in some situations, the "film theory" of gas transfer is wrong in concept. The film theory denies the obvious fact of physical surface water replacement, and is based upon the false supposition that a dissolved oxygen concentration gradient is not present within a well-mixed system - as has been seen, such a concentration gradient is the driving force for diffusion, and must exist everywhere within the unsaturated fluid volume.

A clear distinction must therefore also be made between physically impossible stagnant surface water films and physically real hydrodynamic upper layers of water in a system that is not homogeneously mixed. For example, in a stratified reservoir the whole volume of water is physically or hydrodynamically separated into two distinct regions - the lower region has little opportunity for reaeration because its volume elements never reach the air-water interface. In that case, then, the hydrodynamic situation prevents surface replacement and reaeration is very slow. However, this has to do with the hydraulic properties of the system, and has nothing to do with the film theory.



$$(C_1-C_2) = \Delta C = \text{large}$$

$$\therefore \frac{(\Delta C)}{(\Delta r)} = \text{large}$$

$$\therefore J \cong -D_m \frac{\Delta C}{\Delta r} = \text{large}$$

FIGURE 3

GAS TRANSFER IN TURBULENT WATER

Basic Reaeration Model - Referring now to Figure 4, the familiar basic reaeration equation is shown, and states simply that the rate of change of the dissolved oxygen saturation deficit, $D = (C_s - C)$, is proportional to the deficit at any time. In that model, C_s is the saturation concentration of DO in the water, and C is the momentary average DO concentration in the water. The proportionality constant, K_2, is the reaeration rate coefficient. The saturation deficit, then, is the driving force for reaeration, and is proportional to the oxygen partial pressure difference between the air and water.

The basic reaeration equation has been derived elsewhere from simple first principles, (8) where it has also been shown that K_2, the bulk gas transfer coefficient, is proportional to the rate of surface replacement per unit volume, $(n\frac{A}{V})$, and that the proportionality constant, a, is directly related to the molecular diffusion coefficient, D_m. Thus, the coefficient, a, is a constant for oxygen in clean water at any fixed temperature, but will be a function of temperature and may also be modified by the presence of pollutants.

The rate coefficient, K_2, is what we are after, and what is measured in the field by the tracer method to be described in the following discussions. As indicated in Figure 4, it is a function of the water surface area, A, and the volume, V, and the rate of surface replacement, n, in new surfaces per unit time. However, it should be noted that the ratio (A/V) is properly regarded as the reciprocal of the whole depth of water *only* if mixing is homogeneous in terms of surface replacement. Thus, for example, the average whole depth of water in a stratified reservoir is meaningless as a measure of K_2 or reaeration capacity. In point of fact, it is probable that *most* natural watercourses are not homogeneously mixed, and hence the average depth of flow is not a useful measure of the depth that is effective in reaeration, or of the effective volume.

The purpose of our research, then, has been to define the hydraulic properties that determine K_2 in real streams; this is tantamount to attempting to define the rate of surface replacement in terms of measurable hydraulic properties such as velocity, depth, time of flow, hydraulic gradient, hydraulic radius, flow, roughness, wetted perimenter, etc. This research, and its results, will be described in the following papers, after the basic tool, the tracer procedure, has been outlined.

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in turbulent water:

$$\frac{dC}{dt} = an\frac{A}{V}(C_s - C)$$

or, $$\frac{dD}{dt} = -K_2D$$ (basic reaeration equation)

where

$D = (C_s - C)$ = driving force for gas transfer in turbulent water

$K_2 = an\frac{A}{V}$ = bulk gas transfer coefficient

a = related to diffusion coefficient, D_m

= constant for oxygen in clean water at any one temperature.

$(n\frac{A}{V})$ = cm^2 of new surface exposed per unit time and per unit per unit volume

FIGURE 4

BASIC REAERATION EQUATION

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Relative Gas Transfer Characteristics of Krypton and Oxygen

E. C. Tsivoglou

In a polluted stream, the DO may be simultaneously depleted and replenished by a number of different natural processes, not all of which can be evaluated at any moment. Stream self-purification is quite complex, in that there are too many unknowns. As a result, despite the numerous attempts to do so over the past 60 years, it has not been possible to make accurate and dependable evaluations of stream reaeration capacity on the basis of field measurements of DO. But stream reaeration is the key to self-purification, and the determining factor as regards the necessary degree of waste treatment and resulting costs. Hence, accurate evaluation of stream reaeration capacity has remained a problem of major importance in water pollution control.

This is a classical situation for the use of tracers. Specifically, it is a situation in which a gaseous tracer for oxygen can be used to circumvent, or evade, problems of measurement that have proved to be otherwise insurmountable. In order to apply such a tracer technique successfully we need to:

(a) Select a suitable gaseous tracer for oxygen (one that is not affected by so many additional natural processes);

(b) Be able to observe the gas transfer behavior of the tracer in the stream, under field conditions;

(c) Be able to translate this field tracer information to the corresponding gas transfer behavior of oxygen under the same field conditions.

This discussion is concerned with the problems of selecting a suitable gaseous tracer for oxygen and establishing the necessary relationships regarding the gas transfer behavior of the tracer gas and oxygen, or with problems (a) and (c) above. The following paper will consider the problem of field application, or problem (b) above.

Selection of Tracer

In order to serve as a suitable tracer for oxygen, the tracer gas to be used must meet a number of specific requirements. In particular, the gaseous tracer should be:

(a) Chemically and biologically inert, to the extent possible, at expected temperatures, pressures, etc.

(b) Relatively simple to detect and measure, and not subject to interferences due to the presence of a wide variety of pollutants.

(c) Measurable at low concentrations.

Of course, the tracer should also be relatively easy and safe to use under both laboratory and field conditions, and should not itself bring about any effects on the gas transfer characteristics of oxygen or the hydraulic characteristics of the water.

In regard to the first requirement, the monatomic ("noble") gases such as argon, neon, krypton, xenon and radon seem most likely to be suitable. They have complete outer electron shells and are known to be chemically and biologically inert as a result. Hence, there should be no chemical reaction of the tracer gas with naturally present dissolved materials or with pollutants. Nor should the monatomic gases be subject to extraction or degradation by the aquatic biota, even in the presence of large biological populations of one kind or another. They are different in that oxygen is a diatomic gas, but as outlined earlier (behavior of gas molecules as spheres) this should not reduce their effectiveness as a reaeration tracer. In brief, the monatomic gases should not be affected by the many chemical and biological natural processes that affect dissolved oxygen concentration, and should make excellent reaeration tracers from that standpoint.

If it is to be used effectively as a tracer under real field conditions, the tracer gas must be relatively simple to detect and measure, and measurable at very low concentrations. As a corollary, a little must go a long way in field operations, otherwise problems of preparing and handling large field tracer doses might restrict its usefulness to quite small streams. Clearly, the best solution to these problems of detection and sensitivity is the use of radioactive isotopes of the monatomic gases. For the radioisotopes, detection and measurement techniques and equipment are well-known and readily available, and are simple and direct. They are also highly sensitive, and extremely small concentrations of radioactive material can be measured with great accuracy and dependability. Hence, the most likely gaseous tracer for stream reaeration appears to be a suitable radioactive isotope of one of the inert monatomic gases.

Of the monatomic gases, argon appears to be most nearly like oxygen in molecular characteristics (molecular weight, molecular diameter), and would probably be the most suitable tracer from that standpoint. However, no suitable radioactive isotope of argon is commercially available (either the half-life is too short, or the radioactive emission is not detectable and measurable by commonly used techniques and equipment). Radon could be used, but is not as readily available as others and its decay products, or daughters, are numerous and also radioactive, and subject to chemical and biological uptake and exchange in the stream. A suitable radioactive isotope of xenon is available, and could be used as a reaeration tracer, even though its half-life is a little short for convenience (about 5 days). For various reasons, including availability, detectability, half-life, etc., an isotope of krypton, ^{85}Kr, has appeared to be the best available tracer, and has been selected for application in streams. Its half-life is long (about 10 years), and this could become a disadvantage if the stream reaeration tracer procedure were to become too widely used. In that unlikely event, further consideration should be given to the use of the shorter-lived radioisotope of xenon.

Relative Transfer Properties of Krypton and Oxygen

Having selected krypton-85 as the most suitable tracer for stream reaeration, the next and most important task is to establish, if possible, a firm relationship between the transfer properties of the tracer and oxygen. It is necessary that the tracer faithfully reflect oxygen transfer under a wide variety of conditions, especially of turbulent mixing, temperature, etc.

As shown earlier, reaeration is a direct function of turbulence in terms of surface water replacement, the greater the turbulence the faster the reaeration process. But there is available no procedure by which turbulence, or the rate of surface replacement, can be accurately and independently evaluated. Hence, for example, it is not really possible to reproduce truly identical conditions of turbulent mixing from one experiment to the next, even in the laboratory. On the other hand, for the tracer procedure to be successful, it will be necessary to relate the transfer of krypton-85 to oxygen transfer under the same conditions of turbulence. The solution to this experimental problem has been the trick of simultaneity, by which the problem of measuring turbulence is evaded, rather than directly solved.

Figure 1 shows the laboratory test system used to establish the relative transfer properties of krypton and oxygen. In brief, in any single experiment the transfer of both gases is observed simultaneously in the single turbulent system. As a result, it can be known that for that experiment the transfer of both gases did occur under truly identical conditions of turbulent mixing, even though the actual degree of turbulence is not measured and remains unknown. The turbulence problem is thus avoided.

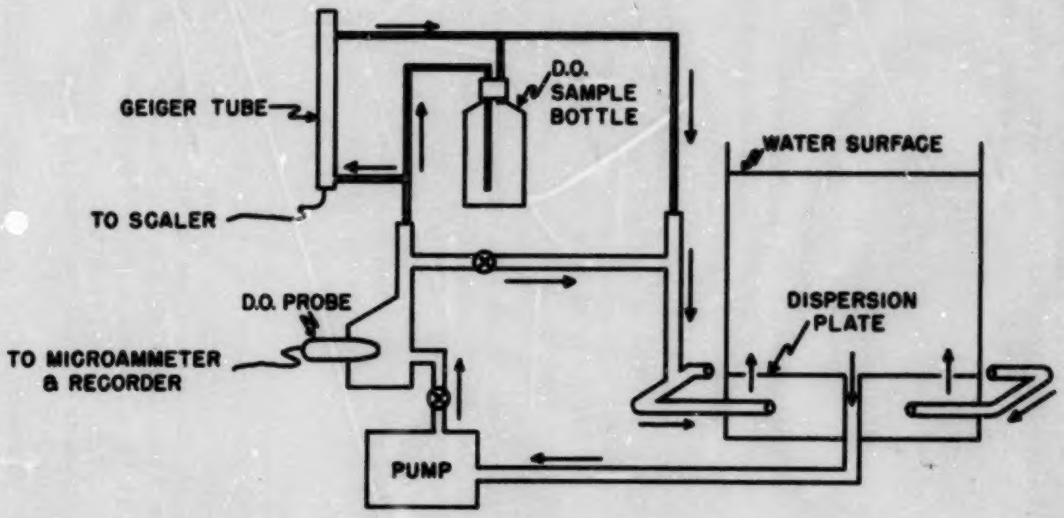
Reactor I consisted of an open cylindrical recirculating water reactor, with external appurtenant equipment as shown in Figure 1. The reactor and most of the external tubing were immersed in a constant temperature bath. Two reactors were used, one of 10-inch and one of 12-inch diameter, each 12 inches deep.

Referring to Figure 1, the test water flowed out of the bottom central reactor outlet, through the pump, and then portions were diverted through a dissolved oxygen (DO) galvanic probe chamber, a

FIGURE I

REACTOR I

SCHEMATIC



water-jacketed flow-through Geiger tube, and a sample bottle (for measurement of DO by the Winkler method). The recombined flow returned to the reactor under a dispersion plate located at the bottom of the reactor vessel, and entered the reactor through a set of peripheral holes in the dispersion plate.

The general test procedure consisted of preparing initial test conditions such that the dissolved oxygen content of the test water was quite low, whereas the concentration of dissolved tracer gas was high. The test then proceeded at a fixed temperature and turbulence condition, and the simultaneous increase of dissolved oxygen concentration and decrease of tracer gas concentration were observed and recorded until the oxygen concentration approached its limiting (saturation) value. Since the water surface was open to the atmosphere, oxygen concentration in the test water approached, for any test, the limit associated with atmospheric oxygen concentrations; the tracer gas water concentration decreased toward zero, as no experimentally significant quantity of tracer gas was present in the atmosphere.

Under these test conditions, oxygen is being absorbed from the atmosphere, whereas the krypton is being desorbed, and these are the directions of transfer that will occur under field conditions. Also, the experimental system lends itself readily to testing under various conditions of temperature, turbulent mixing and depth.

Figure 2 shows the kind of results obtained from a single experiment. Both gas transfer reactions are simple first order, and a K_2 value is readily obtained for each gas. Figure 3 shows a typical set of results from an experiment with radon, one of the other gases tested. From each such test, a single value of the ratio

$$\left(\frac{K_{kr}}{K_{ox}}\right)$$

was obtained. A series of 26 such experiments was conducted, under different conditions of turbulence (different depths and/or recirculation rates), and the results are shown in Figure 4, for the tracer gas krypton-85.

It is evident from Figure 4 that the ratio of K_2 values for krypton-85 and oxygen is constant for the entire range of test conditions. From these and other tests the value of the constant is

$$\left(\frac{K_{kr}}{K_{ox}}\right) = 0.83 \pm 0.04$$

or, the slope of the straight line fitted through the data. A wide range of mixing conditions was tested (K_{ox} from 0.06 to 0.52 per hour), as well as temperatures from 13° to 32°C.

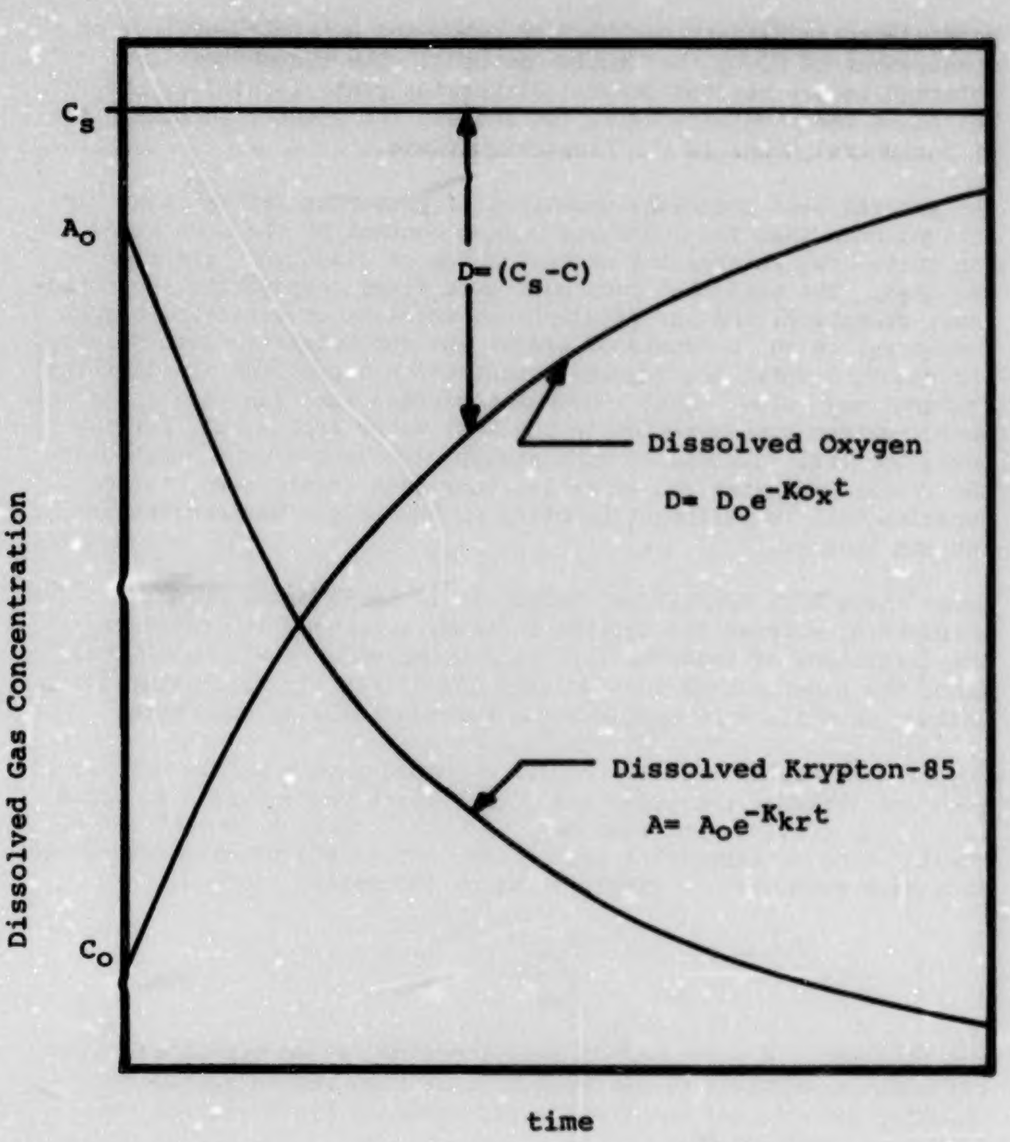


FIGURE 2

OPEN REACTOR EXPERIMENT

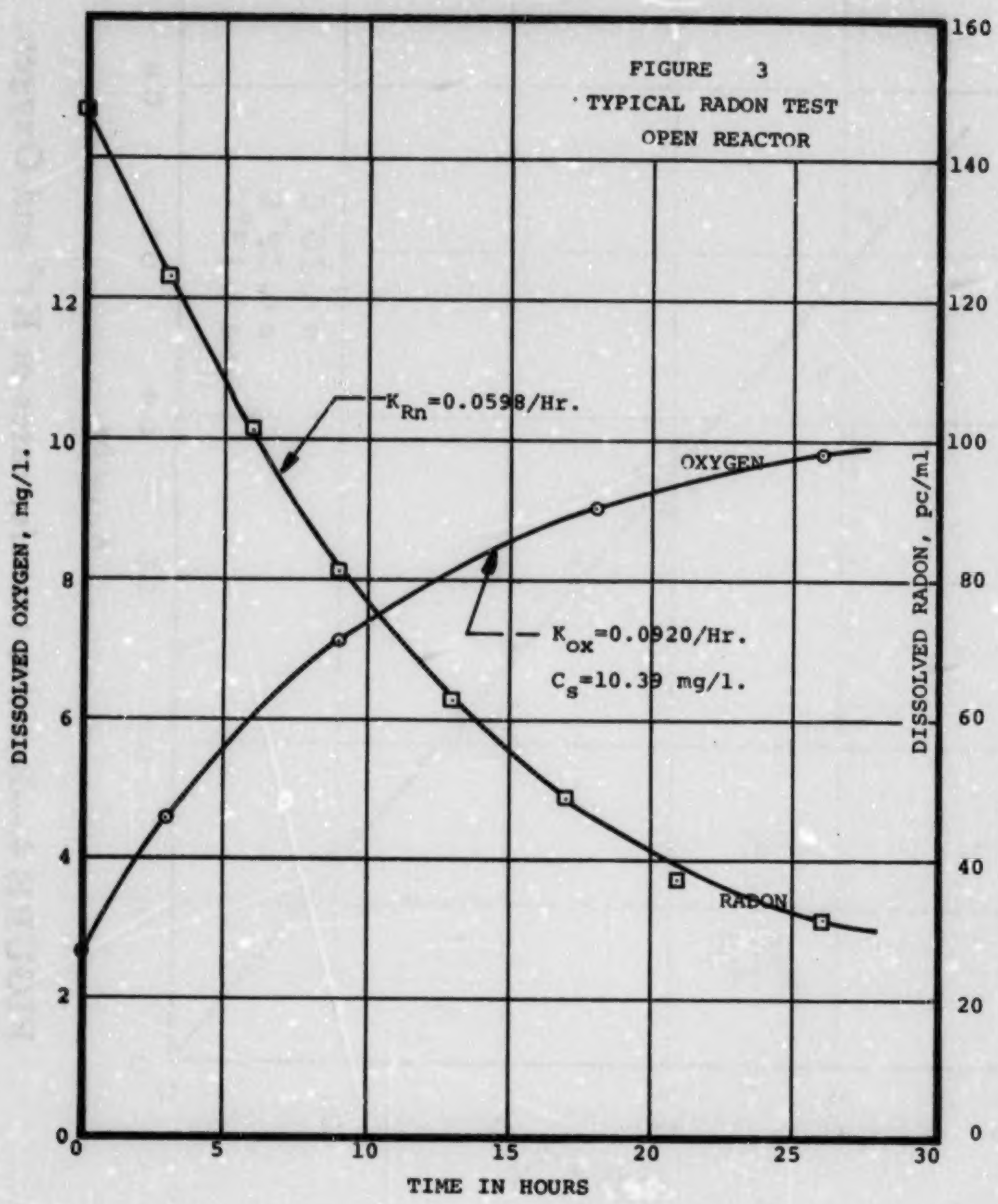


FIGURE 3
TYPICAL RADON TEST
OPEN REACTOR
K_{Rn}=0.0598/Hr.
OXYGEN
K_{ox}=0.0920/Hr.
C_s=10.39 mg/l.
RADON
DISSOLVED OXYGEN, mg/l.
DISSOLVED RADON, pc/ml
TIME IN HOURS
0
2
4
6
8
10
12
0
20
40
60
80
100
120
140
160
0
5
10
15
20
25
30

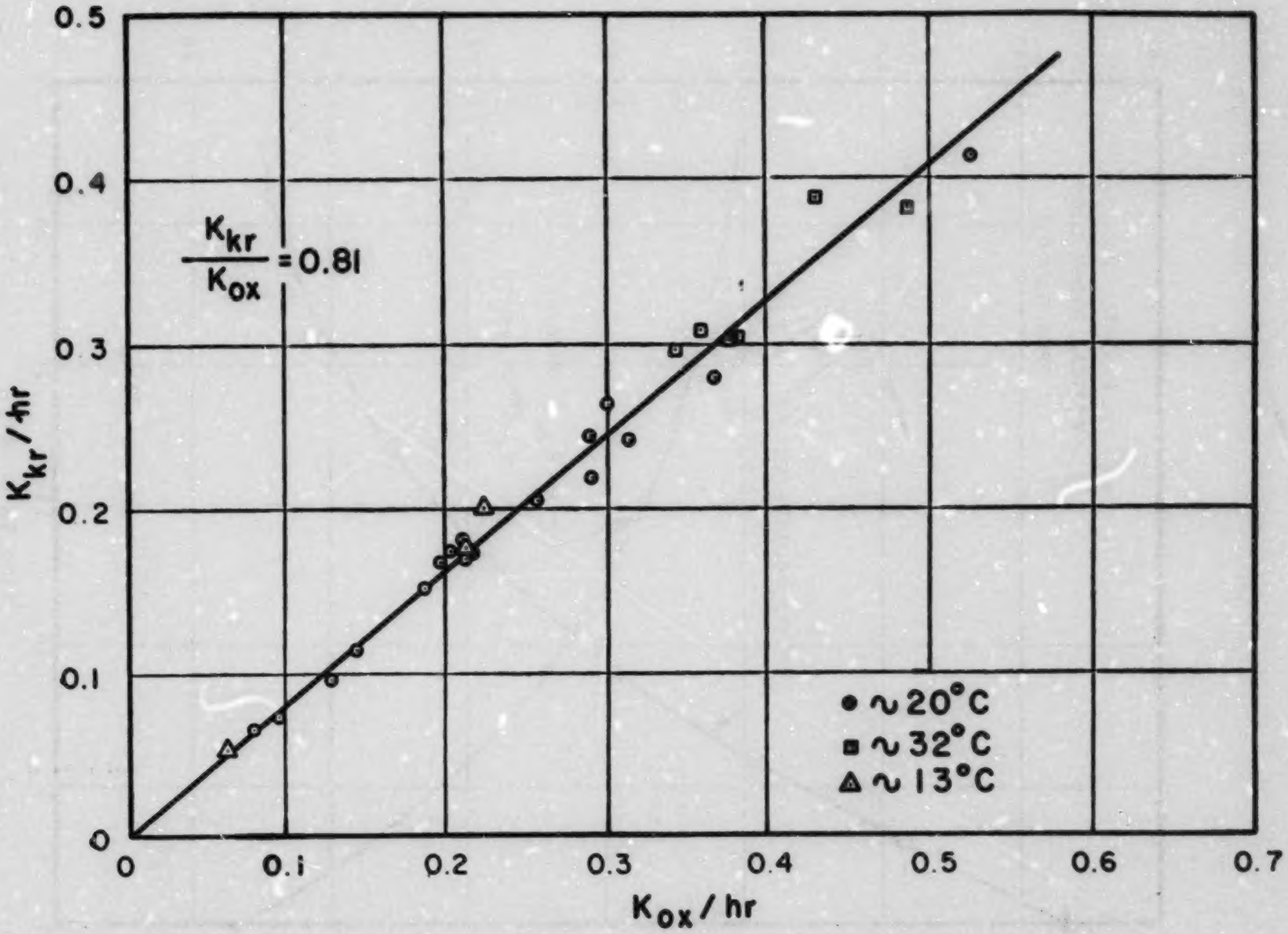


FIGURE 4.—Relative transfer rates of Kr^{85} and Oxygen.

In order to more firmly establish the krypton:oxygen transfer ratio, and for purposes of further investigating the general nature of gas transfer, additional tests with krypton-85 and with other gases were performed in the reactor of Figure 1 and in an entirely different closed reactor in which gas transfer was measured manometrically. The gases tested in both reactors included: hydrogen, helium, nitrogen, oxygen, carbon dioxide, krypton and radon. Figure 5 shows the results of all tests. In each case, the slope of the straight line is the ratio

$$\left(\frac{K_{gas}}{K_{kr}}\right)$$

From the foregoing series of tests performed in FWPCA laboratories, a number of conclusions could be drawn directly:

(1) The relative gas transfer capability of krypton-85 and oxygen, measured as the ratio of reaction rate coefficients, is 0.83 ± 0.04, the transfer of krypton-85 being the slower process.

(2) The krypton:oxygen transfer ratio of 0.83 is not significantly affected by temperature over the range 10°C to 32°C.

(3) The krypton:oxygen transfer ratio of 0.83 is not affected by the degree of turbulent mixing over the wide range studied.

(4) The krypton:oxygen transfer ratio is not affected by the direction in which the gases transfer (into or out of the water).

(5) The krypton:oxygen transfer ratio is not affected by the presence of a broken water surface.

(6) From limited tests with LAS and ABS, the krypton:oxygen transfer ratio is not affected by the presence of such pollutants.

These observations and conclusions have indicated clearly that krypton-85 is a practical field tracer for oxygen and for stream reaeration. Thus, a field-observed value of K_{kr} is directly translatable into a firm value of K_{ox} under the prevailing test conditions of stream flow, temperature, hydraulic properties, etc. It is also clear that the observed value of K_{kr} in any turbulent water system is an accurate and independent measure of turbulent mixing in terms of the rate of surface water replacement.

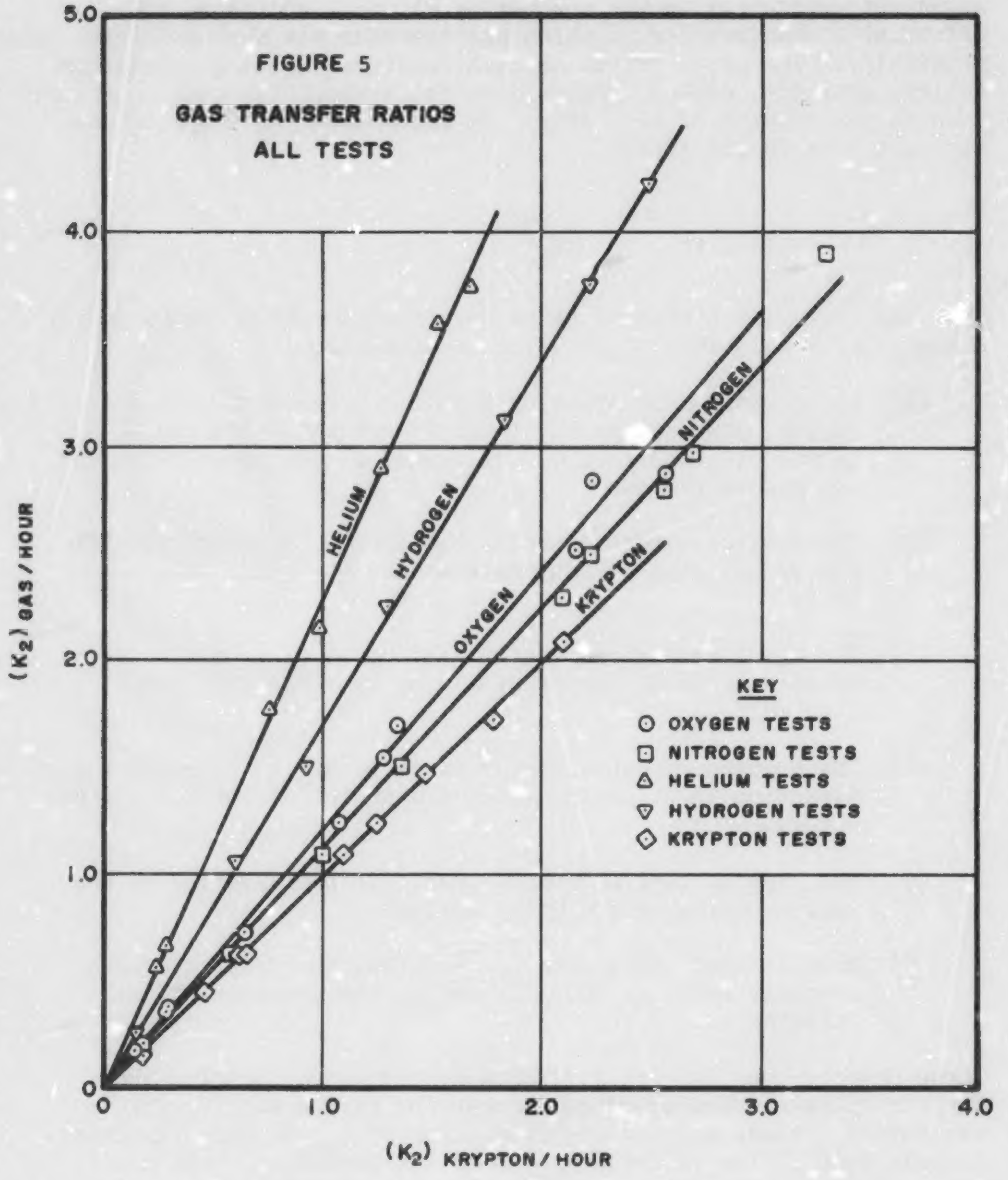


FIGURE 5

GAS TRANSFER RATIOS
ALL TESTS

The tests with other gases led to further conclusions regarding the general nature of the gas transfer process, and thus further demonstrate the usefulness of krypton-85 as a tracer. Figure 6 shows the fundamental relationships involved in gas transfer. Fick's Law and the Basic Reaeration Equation both apply, as outlined earlier. The Stokes-Einstein model for the coefficient of molecular diffusion, D_m, yields quite good numerical results for the transfer of dissolved gases, in terms of observed and predicted values of the diffusivity.

In particular, the last relationship shown in Figure 6 is of practical importance. It can be derived by combining the other models shown above, and was verified experimentally for all of the gases tested. This molecular size rule states simply that for any two different gases, A and B, the ratio of observable values of K_2 is equal to the ratio of diffusivities of the two gases and is equal to the inverse ratio of the molecular diameters. Thus, the larger the gas molecule the less easily it diffuses or transfers from the dissolved to the gaseous phase. Using this relationship, for example, one can predict accurately the diffusivity of a gas, D_m, by comparing either its known molecular diameter or its observed transfer behavior to that of another gas whose diffusivity is known.

The details and all of the results of the experiments referred to here have been published by the FWPCA and are available (1). Having firmly established the validity of using krypton-85 as a field tracer for stream reaeration, the next paper will outline the mathematical basis for field use of the tracer and field procedures.

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$$J = - D_m \frac{dC}{dr}$$ Fick's First Law

and $$\frac{dD}{dt} = - K_2 D$$ Basic Reaeration Equation

and $$D_m = \frac{RT}{6\pi\eta r N_o}$$ (Stokes - Einstein)

$$\frac{(K_2)_A}{(K_2)_B} = \frac{(D_m)_A}{(D_m)_B} = \frac{d_B}{d_A}$$

FIGURE 6

GAS TRANSFER, DIFFUSIVITY AND
MOLECULAR SIZE

Field Tracer Procedures and Mathematical Basis

J. R. Wallace

Fundamental Relationships

The basic phenomena with which we are concerned are diffusion and dispersion. The role played by these physical processes in stream self-purification has been outlined by the previous speakers. My purpose will now be to illustrate how these phenomena can be studied in the field by tracer techniques.

In order to understand the tracer method, it is essential that the equivalence of two processes be made clear. These processes are (1) adsorption of oxygen from the atmosphere into a stream (reaeration) and (2) desorption of a tracer gas from a stream into the atmosphere. In either case the driving force is the concentration deficit, D. In the reaeration process the deficit is the difference between the concentration of oxygen in the water at the point of saturation, C_s, and the oxygen concentration that actually exists, C. Thus,

$$D_{ox} = (C_s - C)_{ox}$$

Likewise, the driving force for a tracer gas (in our studies we use krypton-85) dissolved in the stream is the difference between the concentration of the tracer in the stream and the concentration in the atmosphere. Since the concentration of our tracer gas in the atmosphere is, for all practical purposes, equal to zero, the deficit is simply the concentration in the stream, C_{kr}; that is

$$D_{kr} = C_{kr}$$

As long as the concentration of oxygen in the stream is less than the saturation value, there will be a net movement of oxygen into the stream from the atmosphere, which will tend to reduce the deficit. In a similar manner the net movement of the tracer gas will tend to decrease the tracer concentration in the stream and thereby reduce the driving force. Both of these phenomena can be represented by the general mathematical expression

$$\frac{dD}{dt} = -KD \qquad (1)$$

where K is a proportionality constant which depends on the specific gas under consideration and upon the intensity of turbulent mixing in the stream.

If there were no factors other than turbulent mixing affecting the quantity of dissolved oxygen (DO) in a stream, integration of (1) would provide a means for determining the proportionality constant for oxygen, K_{ox}, through the following relation

$$D = D_o e^{-K_{ox} t} \qquad (2)$$

where D_o is the initial dissolved oxygen deficit (at $t = 0$); and D is the deficit at any later time t. In fact there are many other factors which affect the DO, and therein lies the need for the tracer method.

It has been shown (1) that the ratio of the proportionality constant for oxygen, K_{ox}, to that for krypton, K_{kr}, is constant as long as both gases are subjected to the same conditions of turbulent mixing, i.e.,

$$\left(\frac{K_{kr}}{K_{ox}}\right) = \text{constant} \qquad (3)$$

(A theoretical explanation for equation (3) will be presented in a subsequent lecture.) This relationship between the proportionality constants, together with the fact that the tracer is chemically inert and is not subject to extraction or degradation by aquatic biota, makes it possible to use equation (2) to compute K_{kr} and, through equation (3), to determine the value of K_{ox}.

The tracer method is not quite as simple as it may appear at this point in our discussion, and it is necessary to consider the field procedures in greater detail. Consider two points A and B which lie on a stream, let A be the upstream point, and let a quantity of dissolved krypton-85 be introduced at a point upstream from point A. If this tracer dose were introduced uniformly across the stream cross section, and if there were no vertical or horizontal velocity gradients in the stream causing dispersion and if there were no tributaries causing dilution, then the numerical value of K_{kr} for the reach AB could be obtained from equation (2):

$$\frac{C_A}{C_B} = e^{-K_{kr} t} \qquad (4)$$

where C_A and C_B are the dissolved krypton-85 concentrations at A and B and t is the travel time of flow between A and B. However, we do have dispersion and we do have dilution and they must be taken into account if this method is to be accurate.

Direct measurement of dispersion and dilution is not required; a correction to equation (4) is made possible by using an additional tracer. The second tracer is tritium in the form of tritiated water. Tritiated water is released in the stream simultaneously with the krypton-85. The tritiated water provides an accurate measure of dispersion and dilution. The concentration of tritium decreases between sampling stations because of dispersion and dilution, but, being in the form of tritiated water molecules, tritium is not adsorbed on the stream bed or otherwise lost in any significant amount. Because the tracers are released simultaneously, the dissolved krypton-85 undergoes exactly the same dispersion and dilution as the tritiated water.

Under these test conditions, the observed concentrations of tritium provide an accurate correction for the effects of dispersion and dilution, and hence the decimal fraction of tracer gas remaining at point B is just

$$\frac{\left(\frac{C_{kr}}{C_{tr}}\right)_B}{\left(\frac{C_{kr}}{C_{tr}}\right)_A} = e^{-K_{kr}t} \qquad (5)$$

where (C_{kr}/C_{tr}) A, B are the concentration ratios of krypton-85 and tritium in the samples taken at the time of peak concentration at A and B, and t is the time of flow between the two locations.

If we use only the krypton and tritium tracers we have no way of knowing when the tracers are present at the sampling stations, A and B. Therefore, a third tracer is used to solve this problem. The third tracer is a fluorescent dye, and it performs two functions: it indicates when to sample for the invisible radioactive tracers, and it provides an accurate measure of the time of flow between sampling stations. (If it were not for the fact that the dye is absorbed on the stream bed it could be used to correct for the effects of dispersion and dilution and the tritium tracer would not be required.)

The three tracers are mixed together and are thus injected into the stream simultaneously. Samples are taken from the stream as the dye cloud passes A and B, and the concentration of the krypton and tritium is determined in the laboratory by simultaneously counting the activities in a liquid scintillation counter. Thus, equation (5) is the basis for the field procedures. We measure the krypton-to-tritium ratio at A and at B, we get the time of travel, t, from fluorometer recordings of the dye concentrations at the two stations and plot the results on semi-log paper (see Figure 1). Equation (5) plots as a straight line on semi-log paper when the concentration ratio is plotted on the log scale and time is plotted on the linear scale. The slope of the line is equal to K_{kr}. Knowing the value of K_{kr}, we can obtain the value of K_{ox} (or K_2 as it is usually called) from the known value of the ratio of K_{kr} to K_{ox}.

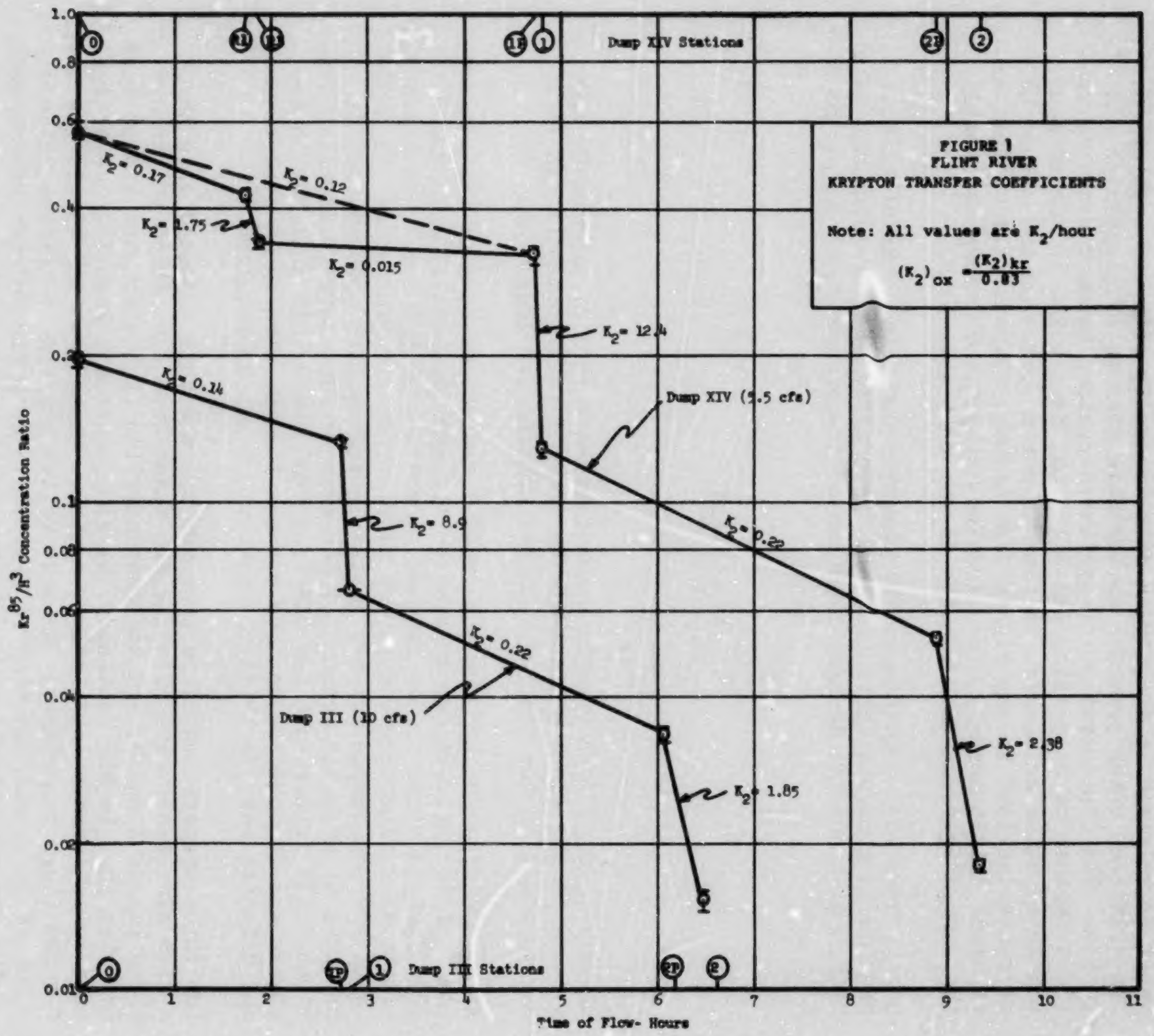

FIGURE 1
FLINT RIVER
KRYPTON TRANSFER COEFFICIENTS

Note: All values are K_2/hour

$(K_2)_{ox} = \frac{(K_2)_{kr}}{0.83}$

The procedure we use for determination of the reaeration coefficient is based on a number of assumptions, and I would like to point these out so that we can all be aware of them. The first assumption is that the nongaseous dispersion tracer (tritium) undergoes only dispersion and dilution, and is not adsorbed on the stream bed or lost in any significant amount. The second assumption is that the tracer gas undergoes dispersion to the same degree as the nongaseous dispersion tracer and, in addition, is lost to the atmosphere; it is not otherwise lost by adsorption or other processes in any significant amount. Third, the tracer gas and the oxygen undergo gas transfer to the same relative extent, and the ratio of their respective K_2 values is not significantly affected by temperature, turbidity, or the presence of pollutants. Fourth, the tracers are released as a truly simultaneous dose, and the concentrations must be measured in the same downstream sample.

Field Procedures

I would now like to make some remarks about the field procedures we use. Most of our studies have been conducted in the Atlanta area, but we also have studied two other rivers, one in Maryland and one in Virginia. The studies have been made, almost entirely, at low flows in the rivers in order to determine the reaeration coefficients for critical flow conditions. The sketch in Figure 2 shows one of the streams that we worked in. The sampling stations are labeled 1, 2, 3, etc. down the stream. The distances between sampling stations on this river were in the range of one to two miles. In all of our field work we have always made at least two measurements for each section of river in order to get a check on the accuracy and the repeatability of our measurements.

A typical field procedure includes making the dose at a point upstream from the sampling stations, setting up the fluorometers at the sampling stations, measuring the stream discharge at each sampling station, and collecting samples from the stream to bring back to the laboratory for analysis.

Let us now consider these steps in greater detail. A typical flow for the Flint River (the river shown in Figure 2) was of the order of 10 to 30 cubic feet per second (cfs). For this stream and others of similar size we used a dose consisting of one liter of dye, one curie of tritium, and 0.5 curies of krypton. Our doses were prepared for us by a commercial firm dealing in radioactive tracers, and the quantities of radioactivity specified by us were normally 1 curie of tritium and 0.5 curie krypton-85, but we observed quite a lot of variation in the quantities that we actually received. In all cases the dose was assayed in the Georgia Tech laboratory before the tracer was released.

The dosing procedure we use is very simple. The dose was delivered to us in a one liter bottle. The bottle is placed in the dosing rig shown in Figure 3. The dosing rig is fabricated from steel channel sections.

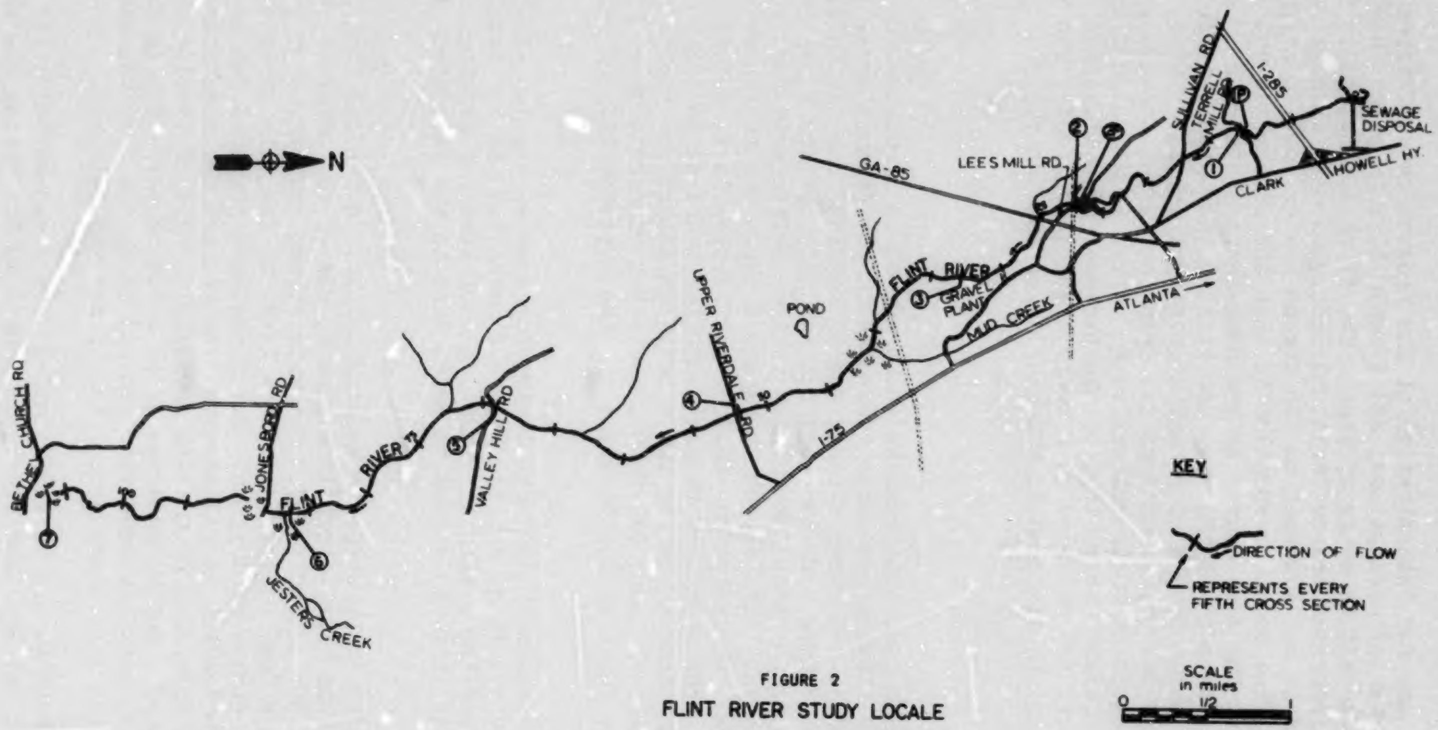

FIGURE 2
FLINT RIVER STUDY LOCALE
VICINITY OF ATLANTA

TRACER RELEASE DEVICE (MANUAL)

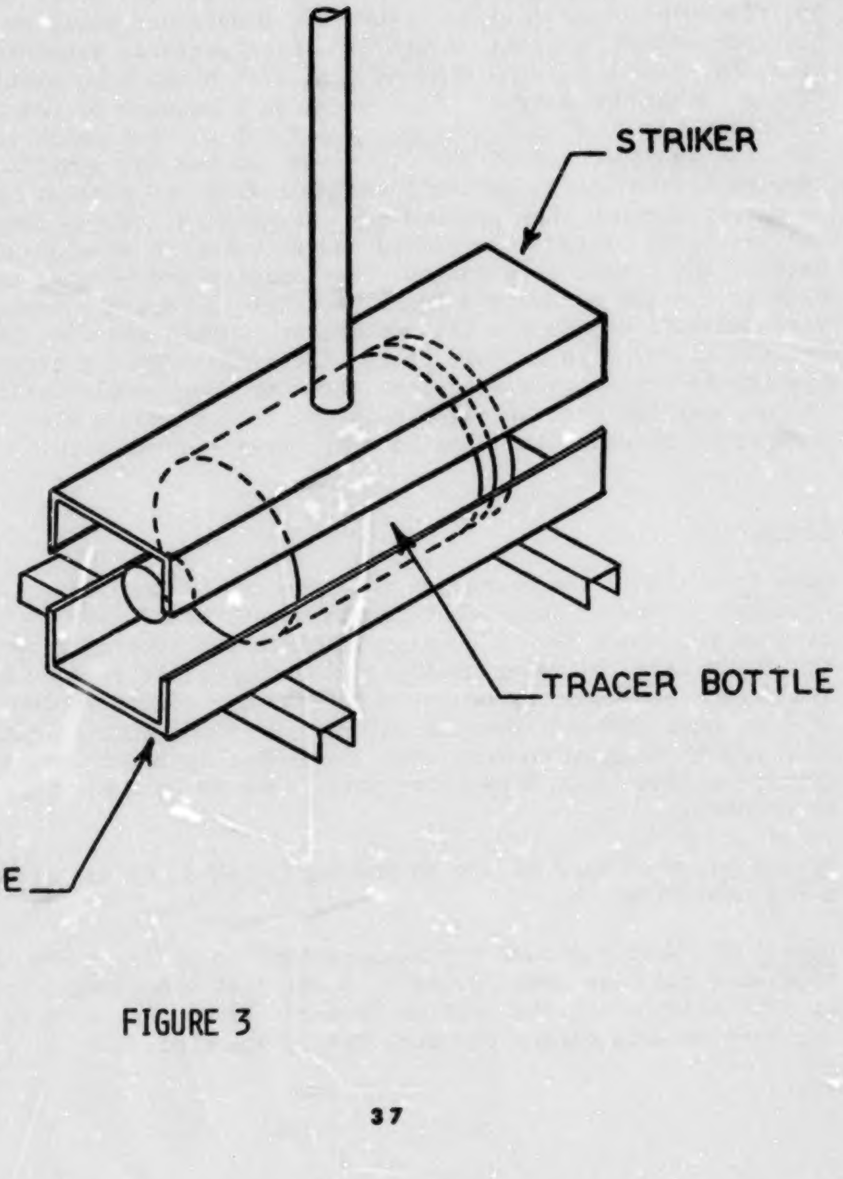

FIGURE 3

The bottle containing the dose is placed between the channels and taped in position. The dose is placed in the stream with the bottom of the rig resting on the bottom of the stream. When the rig is in place, the rod which serves as a handle for the rig is struck with a hammer. This shatters the bottle and provides an instantaneous release of the tracer. In the smaller streams the dye cloud is dispersed rapidly across the stream channel and within a few yards completely fills the channel section.

Fluorometers, instruments which provide a continuous record of the dye concentration in the stream, are placed on the banks of the stream at the sampling stations and put into operation before the tracer material is released. We use three of these instruments, so we were able to set up three sampling stations prior to making the dose. A sketch of the fluorometer is shown in Figure 4. Electrical power to run the fluorometer and the pump is provided by a portable generator. The first thing that is done when we arrive at a sampling station is to take a background sample. This provides a measure of the radioactivity and *fluorescence* that is naturally present at that point in the stream. When the leading edge of the dye cloud reaches the sampling station samples are collected by continually drawing off part of the water that is moving through the fluorometer. We continue taking samples until the dye concentration has peaked and returned to a value equal to about half of the peak concentration. The samples are bottled and shipped back to the lab at Georgia Tech. The bottle caps are designed to prevent the loss of gas and the bottles are marked and the time of collection is noted in a field book. After the sampling procedure at a station is terminated, we select three or four samples which were collected near the concentration peak and send these samples, along with background samples, into the lab for immediate analysis.

Discussion

Question: Could the procedure that you use be used in a lake or estuary? (JRW) I think that the technique would have to be changed because you would have a completely different type of flow system. You don't have the water moving down a channel like we do here, you would have problems trying to follow the dye concentration. I think that we have someone speaking later on/or about work in estuaries and they may be able to comment more knowledgeably about this particular question. But, I do know there would have to be some change in field techniques.

Question: What kind of dye do you use? (JRW) We use rhodamine-WT at a 20% concentration.

Question: Have you done any measurements to indicate how much absorption--how much dye loss you get? I was just wondering if you could get by with eliminating the tritium tracer? (JRW) No, we have not done any work on determining how much dye is absorbed.

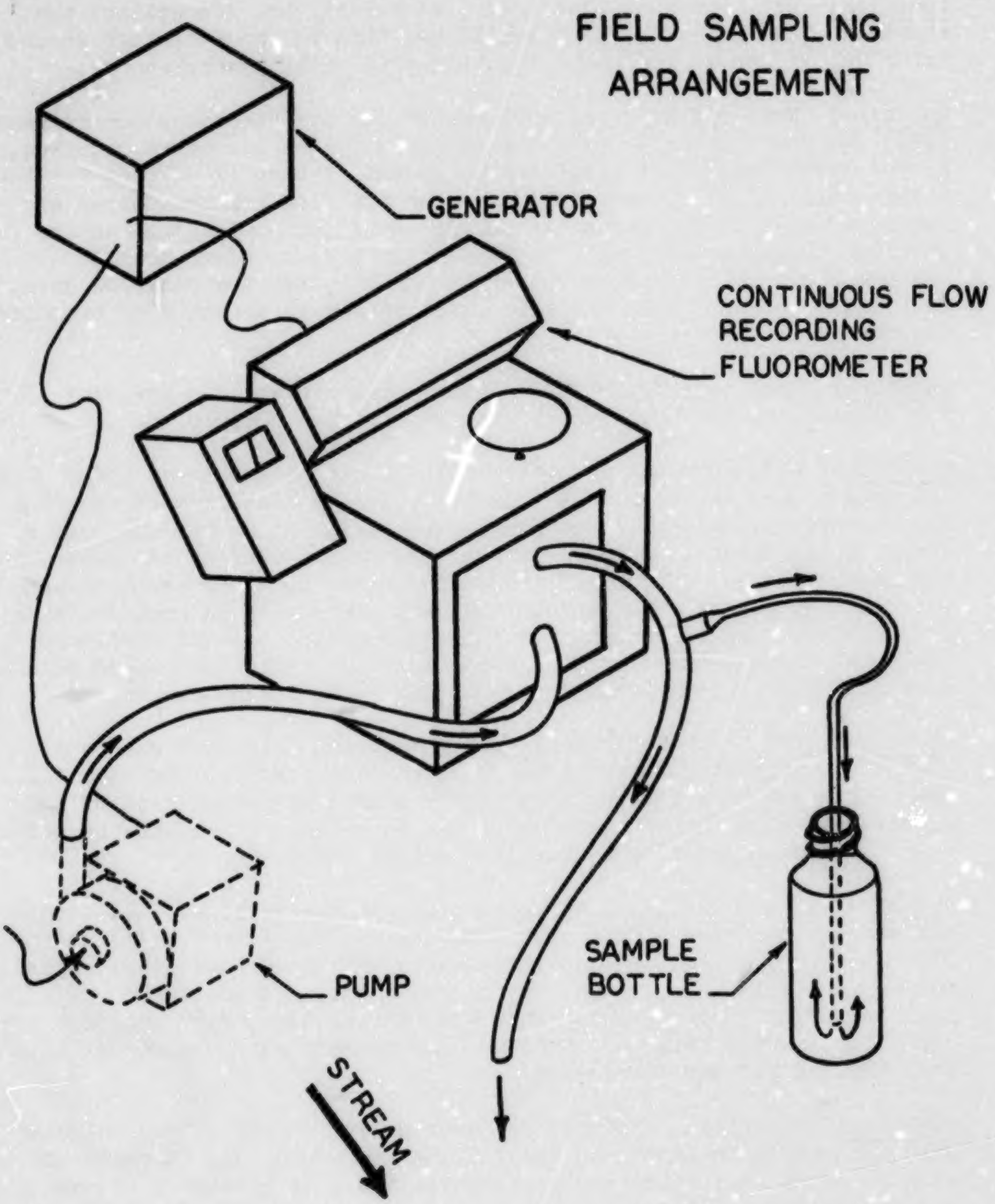
FIELD SAMPLING ARRANGEMENT
GENERATOR
CONTINUOUS FLOW RECORDING FLUOROMETER
PUMP
SAMPLE BOTTLE
STREAM

FIGURE 4

Question: Will you show later, more in detail, how you collect the samples? (JRW) Yes, I think you'll see this in the film that we show later on; it has a good shot of just how we collect samples.

Question: Have you done any analyses of the dye concentration curves? (JRW) We don't normally take the complete dye curve. This requires, as you might imagine, a considerable amount of time in waiting for the entire curve to pass, particularly after the flow has progressed a number of miles. The curve, of course, gets quite elongated and so in order to avoid this large commitment of time and personnel to measure the complete dye curve, we have just taken the peak and down, as I said, to about half the peak which seemed to be the most expedient for our particular purpose.

Question: What is a typical velocity in the streams you work in? (JRW) The velocity is in the order of 1 to $2\frac{1}{2}$ feet per second.

Question: What were the lowest velocities in the streams? (JRW) Well, the lowest were in pools. We took the velocity measurements at a number of intervals along the stream between sampling points and, under these circumstances, we scarcely ever got below 0.9 feet per second in our measurements. Of course, the measurements on these small streams were made by wading. We avoided the deep pools that we couldn't wade in and these were very slow, but they represented a small portion of the total stream, that is, the proportion that would correspond to a very low velocity was a small part of the total stream length.

Question: Where in the channel section did you take your samples? (JRW) Predominantly we would take center line samples. We have taken some for a check at intermediate points across the cross-section and we haven't found that these give us any measurable difference in our results when compared to center line measurements.

You have to be careful not to measure at the center line at one point and over at the edge at the next. Before we locate the intake to the fluorometer we go out and determine where in the cross-section we have our maximum velocities, so we try to get the intake where the peak will be coming through rather than over on the edge where the peak might be somewhat delayed. But, we have checked this and we can't find that we get any difference.

Question: Is tritium absorbed and does pollution affect your studies? (JRW) The question regarding the tritium--we would only be concerned with it if it were selectively absorbed, i.e., if something happens to it that doesn't happen to the water. We have no evidence that this occurs. The tritium is in the form of triated water and as far as we can determine it behaves just like water. Now for the second point you raised concerning the affect of pollutants of one type or another. We have run extensive series of tests with various types of pollutants to see if they affect the ratio, i.e., the proportionality, between oxygen and krypton coefficients. They do not. Nothing that we have been able to find will change this ratio. Certainly the individual values of K_2 are affected, and we'll talk more about this later. The effect of pollutants is on the actual value of K_2, but not on the ratio.

Question: Would you again explain the use of the tritium tracer? (JRW) We take the ratio of krypton to tritium at station B, and divide it by the ratio of krypton to tritium at station A--the tritium is merely reflecting the amount of dilution or dispersion that takes place in the stream. So, it is just an adjustment for any dilutions. If a stream is diluted by 50% then the tritium will be 50% lower in concentration and the same effect will be present on the other tracer--the krypton. By dividing through by the concentration of tritium we simply adjust for any dilution or dispersion.

Question: Do you use a different procedure on large streams? (JRW) This afternoon I think we have one lecture scheduled for discussion of the reaeration studies that we've been conducting on the Chattahoochee River which is the largest stream that we have studied and at that time I'll show you that a different procedure is used for some of these measurements, but basically our procedure is the same, whether we are talking about a small stream, 10 cfs, or a large stream, 2,000 cfs.

Field Hydraulic Studies

J. R. Wallace

D. E. Hicks

Introduction

People working in the field of stream sanitation have for a long number of years tried to develop mathematical models which would predict the value of K_2 for a given set of hydraulic and physical characteristics of the stream. These efforts date back 60 years or so. Many questions arise regarding the available equations, and some of these questions are as follows: First of all how should the hydraulic properties be measured? If you're going to measure the velocity of the stream or the depth of the stream or any cross sectional area characteristic, how frequently along the stream do you have to measure these quantities, and how well do you have to measure them? There are also questions regarding the range of error associated with the equations. There are also questions regarding the effects of the non-uniformity of a channel. If a channel is very non-uniform then I think we can anticipate that many of the existing equations for predicting K_2 will work less well than they would if we had a uniform section of channels to work with. One difficulty in giving a good field test to the previously developed models has been in the difficulty of measuring K_2, and this is one thing that we are hoping to improve during our current study. The purpose of our present study, in addition to the development of the tracer method, is to evaluate the range of the error associated with available mathematical models and to develop, if required, modifications to existing models and/or additional models for predicting reaeration. And, as part of this, we have done detailed hydraulic studies on each of the rivers on which we have conducted our tracer measurements.

In our discussion we will describe in detail how the hydraulic studies were conducted. Our comments will apply specifically to the smaller rivers we have studied, the Flint and South Rivers near Atlanta and the Patuxent River in Maryland. The procedures on a larger river that differ from those on the small rivers will be explained in more detail in a subsequent discussion.

Measurement of Hydraulic Properties

The physical characteristics of interest were the discharge, depth of flow, stream width, flow velocity, hydraulic radius, wetted perimeter, and the slope of the stream. Measurements of hydraulic properties were made at 500 ft. intervals. The distances were measured along the main channel of the stream with a steel tagline. At the end of each 500 ft. interval a stake was driven into the stream bank and marked with the

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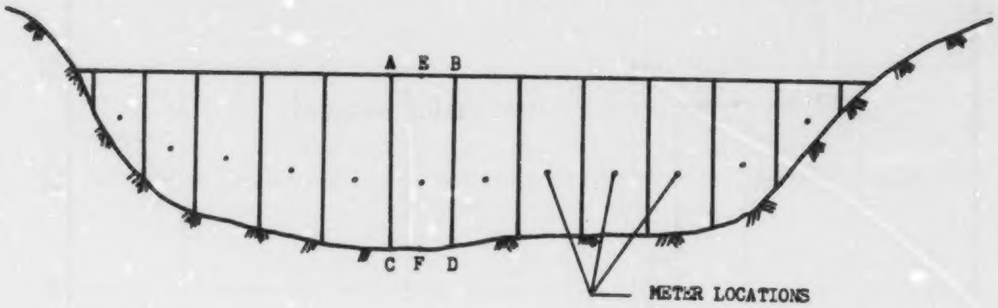
appropriate station number. Standard U.S. Geological Survey stream gauging techniques, with slight modifications, were employed by the field crew. At each station the stream cross section was divided into a number of vertical sections, none of which contained more than 10% of the total flow. An average of 13 vertical sections were taken at each cross section. The width of the cross section was measured with a tagline or steel tape. A depth and velocity were measured for each vertical section by the use of a top-setting wading rod and Price current meter. The Price current meter consists of six conical cups about a vertical axis. Electric contacts driven by the cups close a circuit through a battery and the rod to cause a click for each revolution in headphones worn by the operator. By using a stop watch, the operator is able to record the number of revolutions per time. The meter is calibrated so that the number of revolutions per time corresponds to a velocity. All velocities were measured at 0.6 of the depth in order to obtain a mean velocity in the vertical section. Two men were utilized in measuring distance and making the discharge measurements, although at times one man could work almost as fast as two.

Having obtained these measurements, the other hydraulic values can be calculated. In Figure 1 the cross-section has been divided into vertical sections. For the section ABCD the cross sectional area A is equal to the width AB times the depth EF. The discharge Q for this section is then the velocity measured at 0.6 of the depth, times the cross-section area. With the total cross-section divided into n vertical sections the total discharge is just the summation of the discharges through the vertical sections. The summation from i = 1 to n of V_iA_i.

The Flint River rises in southwest part of Atlanta and flows southward. A reach extending from the Flint River Sewage Treatment Plant, located immediately south of the Atlanta Airport, to a point 9.9 miles downstream was selected for study. The upper two miles is characterized by alternating riffles and pools. In this section are two mill ponds followed by dams approximately 12 feet in height. The remainder of the reach is characterized by a highly variable cross section with quite a bit of debris in the river. In addition, there are two sections, one about a mile in length and the other about 1-½ miles in length, which flow in multiple channels so that no "typical cross-section" can be used to describe these reaches. One reach about 0.3 of a mile long is a marsh where no valid velocity or cross-section measurements can be made. The average depths range from about 0.7 of a ft. in the upstream sections to 1.6 ft. in the lower reaches.

In Figure 2 a plot of flow versus distance illustrate the points at which tributary flows enter the Flint. The Flint River Treatment Plant contributes about 4 cfs, an unnamed stream about 8 cfs, Mud Creek about 2 cfs, and Jesters Creek about 5 cfs. This creates a maximum flow of slightly less than 30 cfs in the lower reach. Figure 3 shows the locations of the two dams and the irregular slopes in the upper reach, and the small slope in the lower section.



$$A_{ABCD} = (AB)(EF) \quad (1)$$

$$Q_{ABCD} = V_{ABCD}A_{ABCD} \quad (2)$$

$$Q_{Total} = \sum_{i=1}^{n} V_iA_i \quad (3)$$

FIGURE I

DISCHARGE MEASUREMENTS

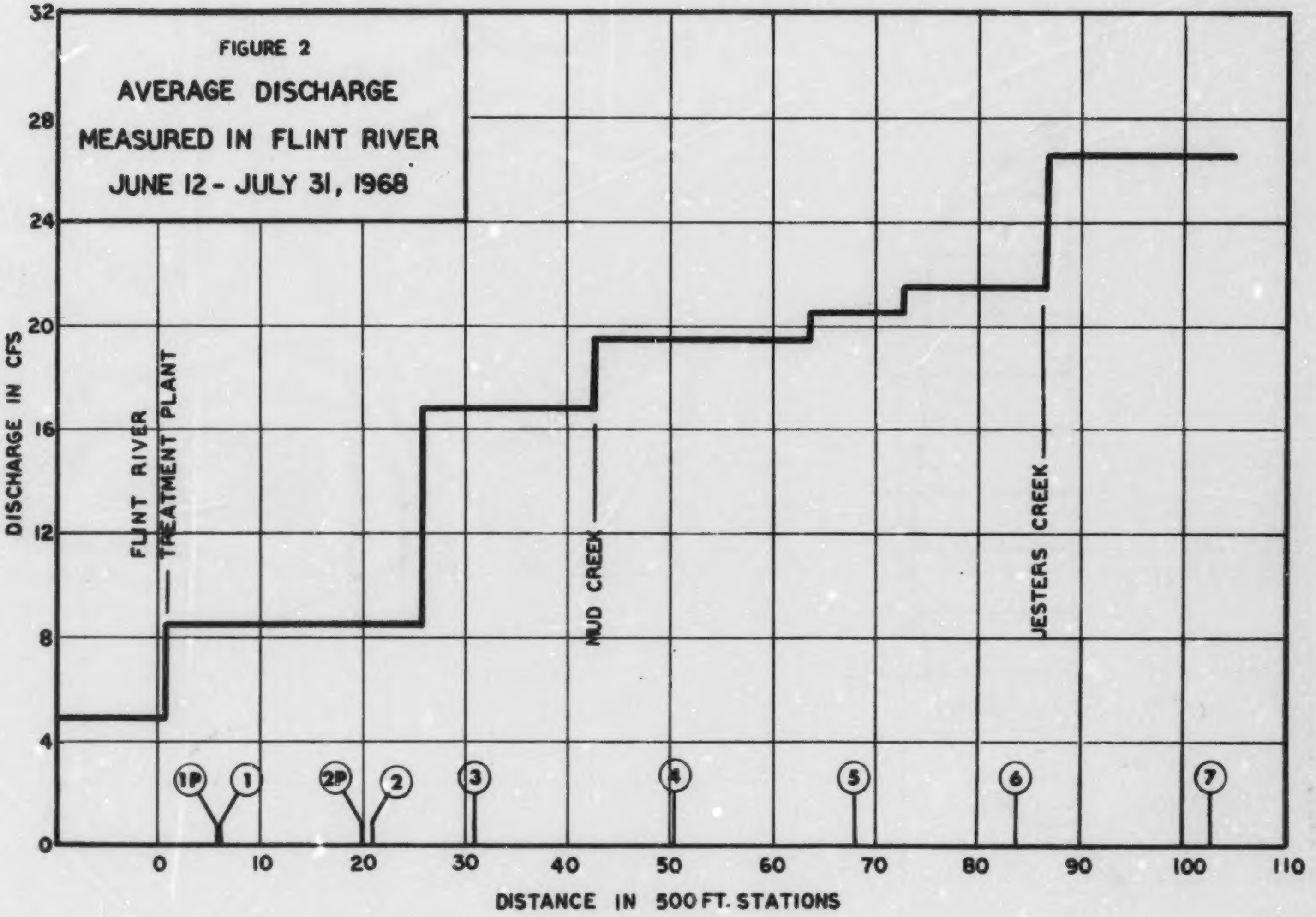


FIGURE 2

AVERAGE DISCHARGE
MEASURED IN FLINT RIVER
JUNE 12 - JULY 31, 1968

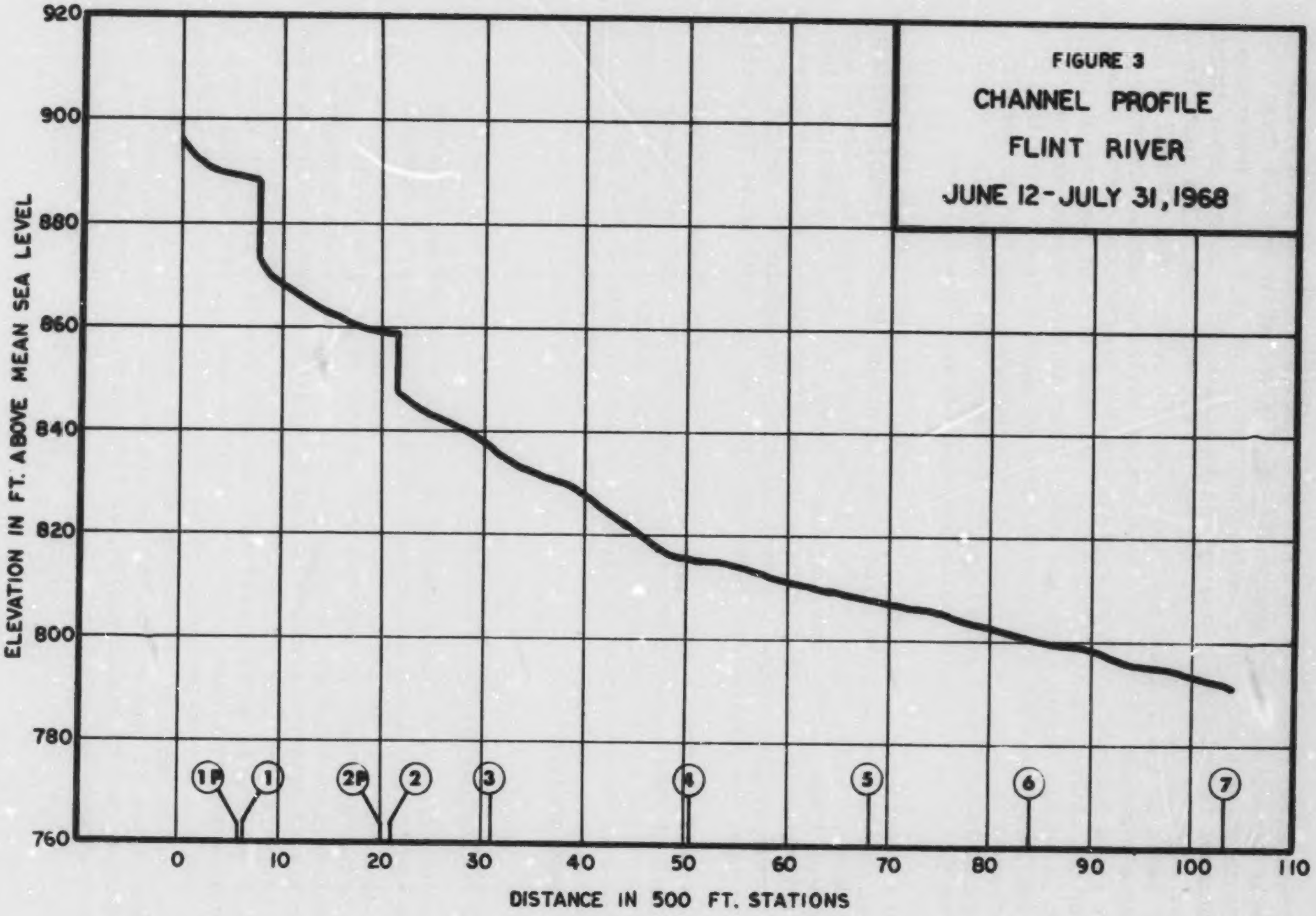


FIGURE 3

CHANNEL PROFILE

FLINT RIVER

JUNE 12-JULY 31, 1968

The South River originates in southwest Atlanta and flows in a southeasterly direction. The portion included in the study extends from the South River Sewage Treatment Plant to a point 18.3 miles downstream. The channel is relatively uniform. A typical section in the upper third of the reach is 30-40 ft. wide and one to two feet deep. The depth to width ratio is fairly constant throughout the reach. The channel is relatively straight with high, practically vertical banks. There are a few pools followed by short rapids, but the predominant characteristic of the channel is its uniformity. The most unusual feature of the South is Panola Shoals where the water flows in thin sheets over a granite formation with an elevation drop of about 8 feet. This drop is preceded by a pool with widths of approximately 100 feet and depths of six to seven feet. Velocities in the South River ranged from about 0.8 to 1.6 ft./sec., with depths averaging from 1.2 to 2.1 feet.

Figure 4 shows the discharge of the South. The contributors to flow in the South River are the South River Treatment Plant, 20 cfs, Intrenchment Creek, 25 cfs, Sugar Creek, 15 cfs, Shoal Creek, 12 cfs, Snapfinger Treatment Plant, 3 cfs, and Snapfinger Creek, 12 cfs. The maximum flow is just slightly less than 130 cfs in the lower reach. Figure 5 shows the stream profile. Note the rapid sections throughout the length of the stream, the location of Panola Shoals, and the pool above the Shoals.

In the fall of 1969 we conducted studies jointly with the Maryland Department of Water Resources on the Patuxent River. This stream is characterized by its uniformity. During the low flow study period depth ranged from 4 to 6 inches in riffle sections to about 2 feet in the deeper pools. Velocities were slow for most part and very uniform over the reach studied.

Adjustment of Hydraulic Properties

Since the field and tracer studies were conducted during different flow conditions, a method had to be devised to adjust the measured physical characteristics to the actual physical characteristics during the tracer studies. We had measured several miles of hydraulic characteristics at 500 feet intervals, and in order to relate hydraulic property to reaeration values we had to have an estimate of the hydraulic properties as they existed at the time of the tracer studies. The method we used is quite simple and straight forward. Figure 6 shows typical cross sectional conditions, with A_1 and P_1, respectively, the area and the wetted perimeter at the time we did the hydraulic studies. Then, on the basis of the Manning equation, equation (1), we adjusted these values to take account of the changes in depth, area, wetted perimeter, and hydraulic radius, as they existed at the time of the tracer study:

$$Q_1 = \frac{1.49}{M_1} A_1 R_1^{2/3} S_1^{1/2} \qquad (1)$$

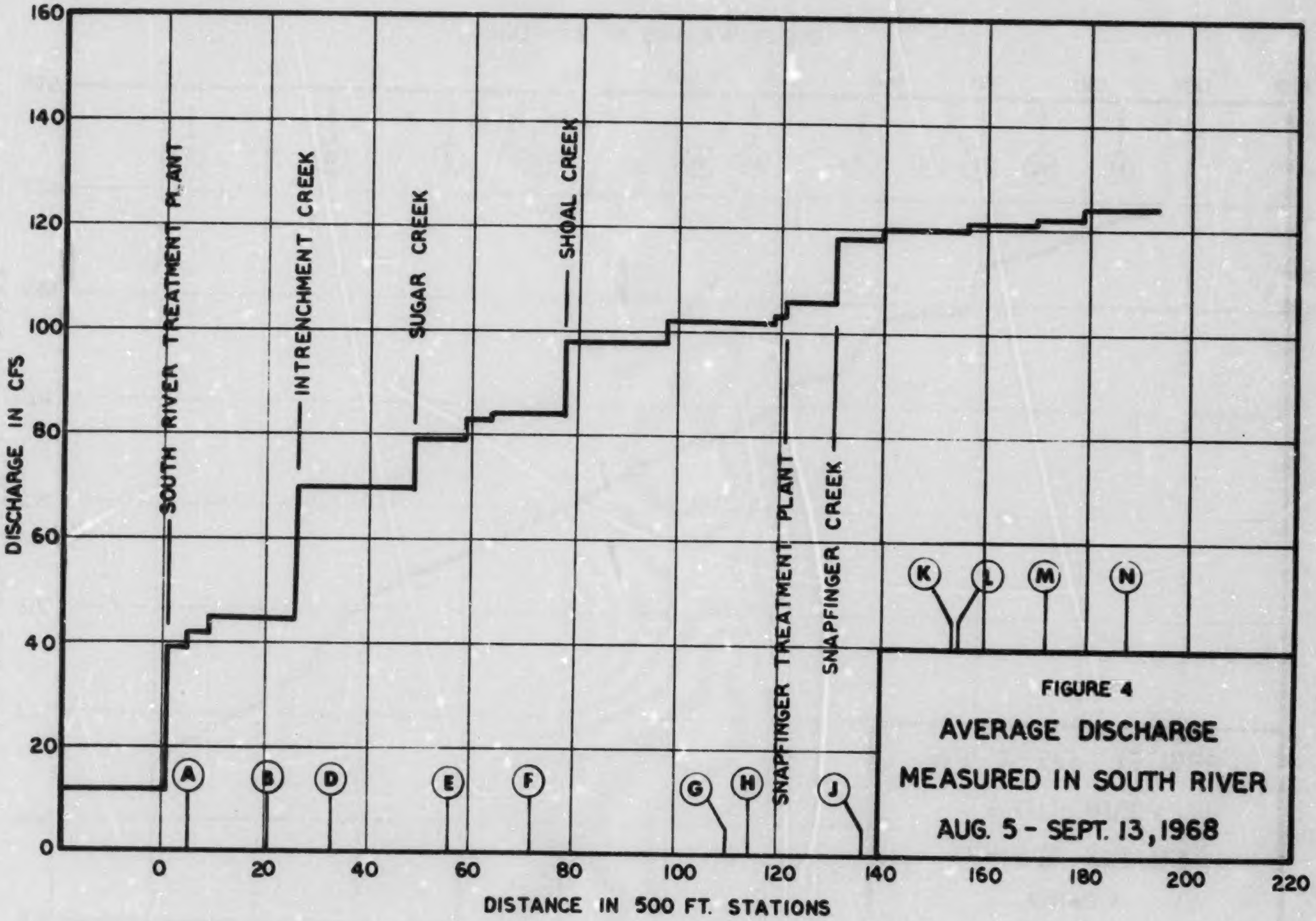


FIGURE 4
AVERAGE DISCHARGE
MEASURED IN SOUTH RIVER
AUG. 5 - SEPT. 13, 1968

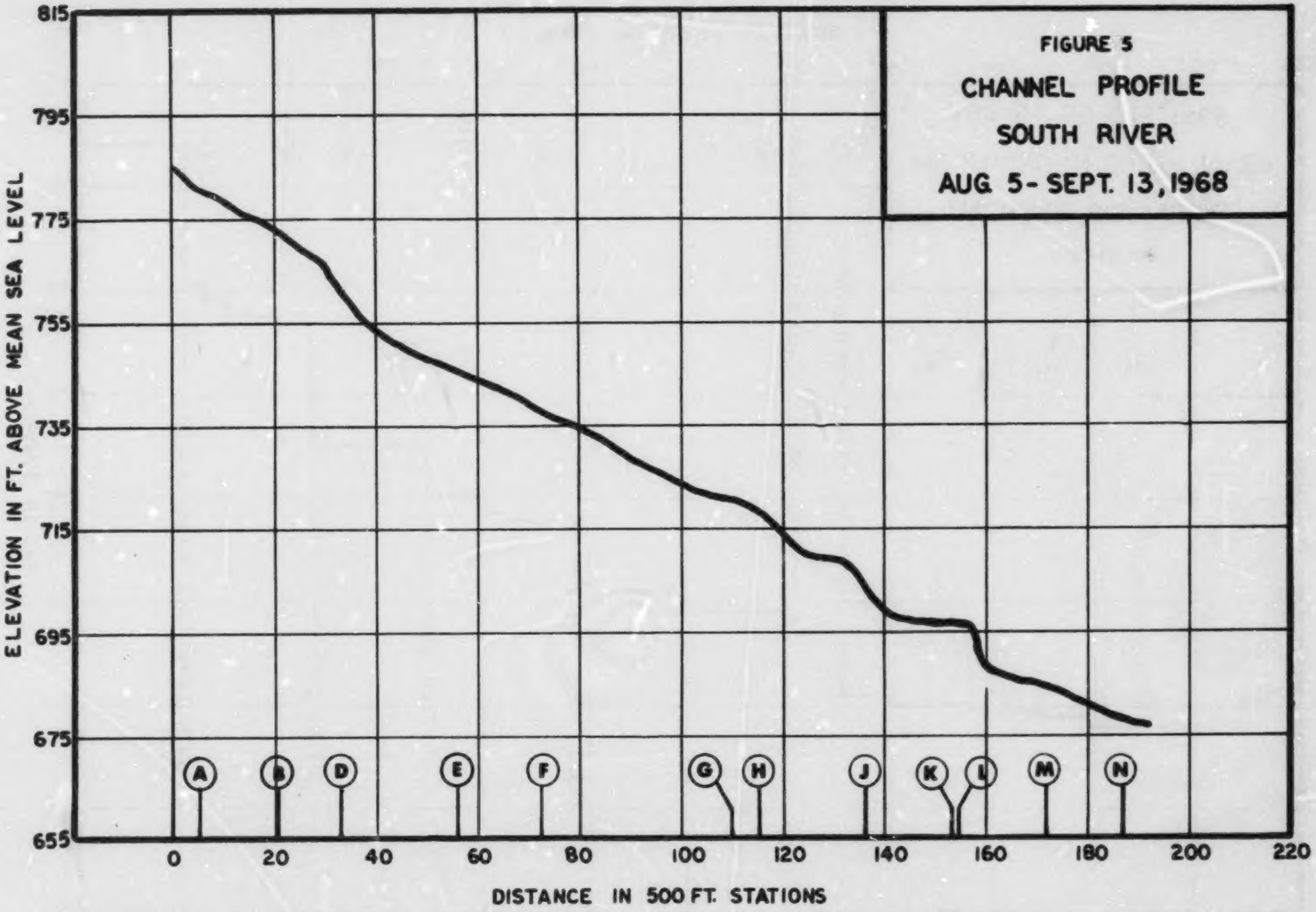
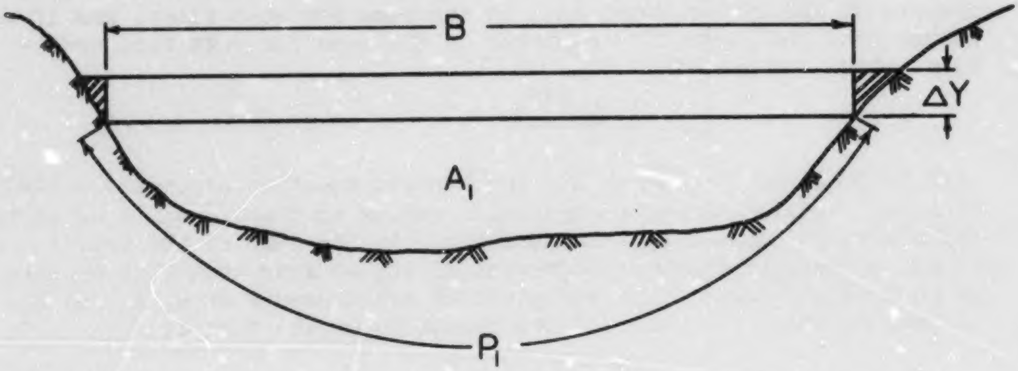


FIGURE 5

CHANNEL PROFILE

SOUTH RIVER

AUG. 5 - SEPT. 13, 1968



$$Q_1 = \frac{1.49}{n_1} A_1 R_1^{2/3} S_1^{1/2} \qquad (1)$$

$$A_2 = A_1 + B\Delta Y \qquad (2)$$

$$P_2 = P_1 + 2\Delta Y \qquad (3)$$

FIGURE 6--METHOD OF ADJUSTING FLOW

where Q is the discharge (cfs), R is the hydraulic radius (A/P), n is the roughness factor and S is the slope of the energy grade line. The subscript 1 represents conditions as originally measured and subscript 2 represents conditions during the tracer release.

We are also saying that the perimeter is equal to the perimeter that we originally measured, plus 2 times ΔY. The implication of these assumptions is that we are disregarding the section of the diagram that is shown cross-hatched. Having decided that we are going to use the Manning equation, then if we assume that the n value, which is a measure of the stream roughness, is the same for both flows, and if we assume that the slope of the stream is the same for both flow conditions, then the quantity

$$C = \frac{(1.49S^{1/2})}{n}$$

will be constant throughout and the Manning equation becomes $Q = CAR^{2/3}$, that is, the area times the hydraulic radius to the 2/3rds is equal to the discharge divided by this constant. At the time of the tracer study, we measure Q, so we know what Q_2 is, we know what C is because we have solved for it from our previous measurements of Q, A, and R_1. We know A_2 and R_2 in terms of the change in depth, that is

$$A_2 = A_1 + B\Delta Y$$

and $$P_2 = P_1 + Z\Delta Y$$

thus $$Q_2 = C(A_1 + B\Delta Y)\left(\frac{A_1 + B\Delta Y}{P_1 + 2\Delta Y}\right)^{2/3} \qquad (2)$$

The only unknown in equation (2) is ΔY. We solve for ΔY and then use ΔY to compute the value of the hydraulic parameters that existed at the time of the tracers study. In this way, we have an adjusted value of the hydraulic parameters to be used for correlation with the measured reaeration rates.

Discussion

Question: What was the magnitude of the variation in flow conditions between the time of the hydraulic studies and the tracer studies? (JRW) The variation between the conditions at the time we did the hydraulic study and at the time of the tracers study were not great. By not great, I mean they were in the order of 10% of the discharge at a given point. That's a typical figure. For example, discharge at a station may have varied from 60 to 66 or 50 to 55 cfs between the times of different parts of the study.

Question: Did you experience variations of flow during the study? (JRW) We did the studies during the summer and early fall, when the streams were at their seasonal low, and if we had any sort of rainfall event we did not make any additional measurements until the stream had receded to its base flow level. We always did this in order to eliminate a large variation during the hydraulic study. The response time for each stream, i.e., the time required for the stream to rise and fall, is short enough so that it did not present any problem if it rained one day. We normally did not have to wait more than one additional day for the stream to go back down to its normal flow.

Question: Your adjustment procedures could lead to errors if the assumption you made about the vertical banks is not valid. Please comment on this. (JRW) You are right. This could be very misleading. We made the adjustment in this way, of course, with the knowledge of what conditions in this particular stream are, and they do follow the assumption pretty closely. I think I have another slide here that shows the typical section and you can see the vertical banks. There are some exceptions, of course, and there are always going to be places where the stream spreads out. We have checked our adjustment procedure and find that the adjustments were always in the range of the accuracy of our measurement, probably about 10%. So we feel like it is better than simply measuring the hydraulic parameters one time and then a month later coming back and assuming that they still exist. It is the only way I know to adjust a large length of river when we are interested in a large number, 100 to 200, of intermediate stations.

Question: Why not estimate the slopes of the banks at various sections and use this slope in your adjustment procedures? (JRW) I think this would give us added accuracy. We didn't do it primarily because we found that we could reproduce conditions within our accuracy in measuring it again. All of us have walked these rivers from one end to another so we have a pretty good feel for the physical situation, but in any case you would have to have such knowledge before you apply any such adjustment technique, and you might have to make some modification in it to meet most of the situations you would find elsewhere.

Question: Was there much variation in discharge as you move downstream? (JRW) Yes, quite a considerable variation. You probably noticed on the plot of discharge versus distance that the South River Treatment Plant puts in an amount of discharge that is probably twice as great as the discharge upstream from the plant and when you go on downstream you pick up three or four other sewage treatment plants. We did take the variation into account when we did our studies.

Laboratory Procedures

R. J. Velten

General Discussion

Before discussing the general techniques used in the laboratory procedures, let me briefly describe what we are measuring.

Tritium is the heaviest isotope of hydrogen and has a mass of 3 atomic mass units. It decays by beta emission with a half-life of 12.26 years to helium-3. Its maximum beta energy is 18.6 kilo electron volts (kev). Tritium can exist in any physical or chemical state in which hydrogen can manifest itself.

Krypton-85, an inert gas, decays both by beta and gamma emission to rubidium. The beta emissions occur 99.6% of the time with a maximum beta energy of 670 kev. The 0.4% abundant gamma emissions are characterized by the 512 kev photopeak.

Because of its extremely low energy, tritium cannot be measured by the usual laboratory counting instruments. Gas counting of the tritium gas as well as liquid scintillation counting as tritiated water are the only plausible methods of measurements. Tritium, when used in the determination of reaeration capacity, is in the chemical form of water (HTO or T_2O) and hence liquid scintillation counting offers the best choice from the standpoint of ease of sample preparation combined with detector efficiency and background.

Krypton-85, on the other hand, can be measured by both beta and gamma counting. However, because of the low gamma abundance, the sensitivity of gamma counting is poor to such an extent that the concentrations of krypton that are usually encountered in the tracer reaeration procedure cannot be measured. Beta counting is the only choice. The dissolved krypton could be purged from the solution and counted, using a gas counter. The high counter background plus the uncertainty that all the krypton has been purged from solution makes this technique less reliable and less sensitive.

Liquid scintillation counting thus becomes the only choice and is especially suited for measuring both radionuclides simultaneously. This counting technique becomes practical for simultaneous measurements of two radionuclides whenever the beta energies differ by a factor of five or more.

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Sample Preparation

Some care must be exercised in the preparation of the samples. Since Kr-85 is an inert gas dissolved in the water, reasonable efforts should be taken to prevent the loss of the gas from solution while preparing the sample for counting. In order to minimize the loss of gas, a transfer rig, as shown in Figure 1, is used. It is thought that by applying gentle pressure to the whole sample and collecting the aliquot from the bottom of the bottle is a better technique in aliquoting than the usual suction-type pipetting technique. The aliquot is then transferred to the scintillation vial and the scintillation solution gently added until the vial is completely filled. The bottle is tightly capped and the liquid phases mixed by swirling, not shaking. However, past experience has shown that the hydraulic pressure created when closing an overfilled vial has occasionally broken out the bottom of the vial. This may be especially important if vials with polyseal caps are used.

The sample is then placed in the counter. If the counter is equipped with a refrigerated system, counting at 4°C is recommended to retard or eliminate any loss of the gas from solution. This is about the lowest temperature where the scintillation mixture will not freeze. If the counting instrument is not equipped with some cooling system, then the instrument should be located in a controlled temperature room. This is to prevent any expansion in the scintillation mixture because of increases in room temperatures. These increases will expand the solutions and samples may be lost be breakage. This will necessitate cleaning the entire interior of the instrument. In preparing the samples, the scintillation solution should not be exposed to fluorescent lighting. This light, as well as sunlight, excites the solution and several hours are required for this excitation to decay before counting can be initiated. Incandescent lighting is satisfactory although this should be minimized.

Instrument Set-up

The setting up of the liquid scintillation counters varies with manufacturers. Thus, I will describe the instrument with which I am more familiar and have used for most of our reaeration studies. This instrument is the Packard Model 3320. The analyzer section of this instrument has three channels, each with its own high voltage supply (gain) and discriminators (lower and upper). To set up one channel for one radionuclide, a discussion on the use of the discriminators and gain control is warranted. Let us assume that a sample has been placed in the detector and the gain is set at some arbitrary low setting. Let us set the lower discriminator at setting 000 and the upper at 050, a difference of 50, and then count the sample and record the count rate. Now, let us increase both the lower and upper discriminator by 50; that is, 050-100 and then repeat and record the count. Continue this increment until no counts are observed. Plot the observed count rate versus the energy

PRESSURE PIPETTE

(NOT TO SCALE)

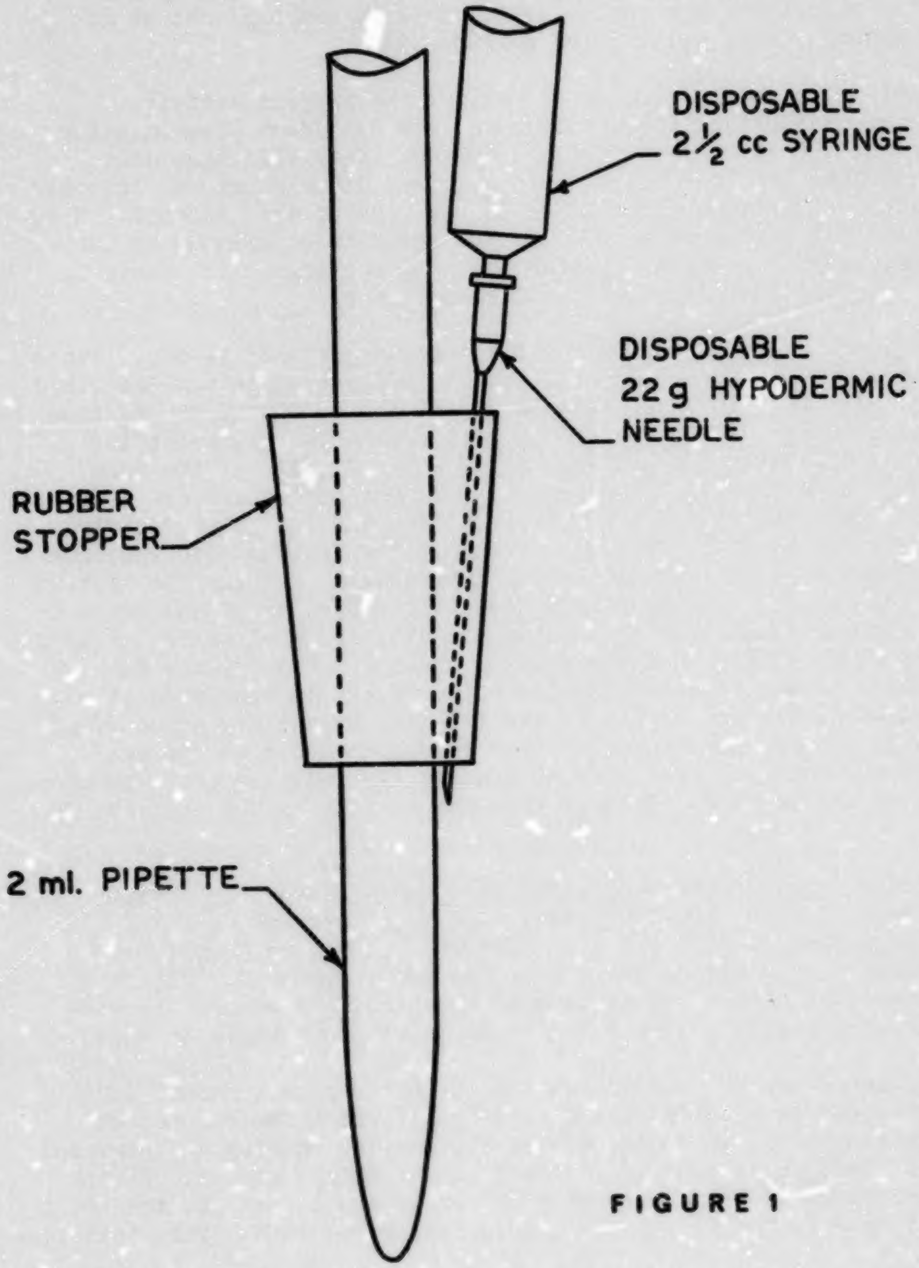


FIGURE 1

band and a curve similar to Curve A, Figure 2, will be described. In this case, the count rate ceases in the energy band 250-300. If the gain is then increased, and the counting repeated, Curve A will take the shape of Curve B. If we further increase the gain, Curve C will be described, and finally if the gain is further increased, the upper edge of Curve C will be beyond the range of the upper discriminator. It can then be seen that, by adjusting the gain control, the dynamic range of the discriminators can be used. This is exactly what we do with each radionuclide, tritium and krypton-85.

First we set the channel which is to monitor the krypton activity. Place a krypton sample in the detector and set the lower discriminator to monitor the energy band at setting 1000-00. This will then monitor any pulse greater than the voltage of the lower discriminators. Increase the gain until significant counts due to the krypton are observed. Now, set the lower discriminator to 000 and the upper discriminator to 1000. What was actually done was to stretch the beta spectrum, like Curve C, Figure 2, over the full range of the discriminator.

Now, the channel which is to monitor the tritium activity is set. Place the tritium sample into the detector and the process which was described above repeated. In this case, because of the low energy of the tritium beta emission, there may not be enough gain available to stretch the spectrum to a discriminator setting of 1000-00. If this is the case, decrease the lower discriminator until counts due to tritium are observed. Now set the upper discriminator at this setting and the lower discriminator is returned to 000. This now stretches the tritium beta spectrum over the widest range attainable by the instrument. Keeping the tritium sample in the detector, increase the lower discriminator of the krypton channel until all counts due to tritium are discriminated against. This then is the final setting for the krypton channel. Now replace the krypton sample in the detector and lower the upper discriminator of the tritium channel until the counts in the tritium channel are about 5% of the counts in the krypton channel. This is about as far as one can resolve the two radionuclides without drastically reducing the detection efficiency of the tritium. This is then the setting of the tritium channel.

If the tritium channel were subdivided into energy bands of 50 units (that is, 0-50, 50-100, 100-150, etc.), and the count rate plotted against energy band, a plot which is described in Figure 3 will be observed, although it would be the sum of the two components. Figure 3 shows the resolved effects of tritium and krypton. If we did likewise with the krypton channel, a curve as shown in Figure 4 would be shown.

The third channel can be used to monitor the external standard. The external standard is a solid radioactive "pill" which is transported next to the sample and which irradiates the samples causing an increased count rate. This count rate is recorded in the third channel. If the physical and chemical properties of each sample are identical, the count rate caused by this "pill" will be statistically the same. This technique

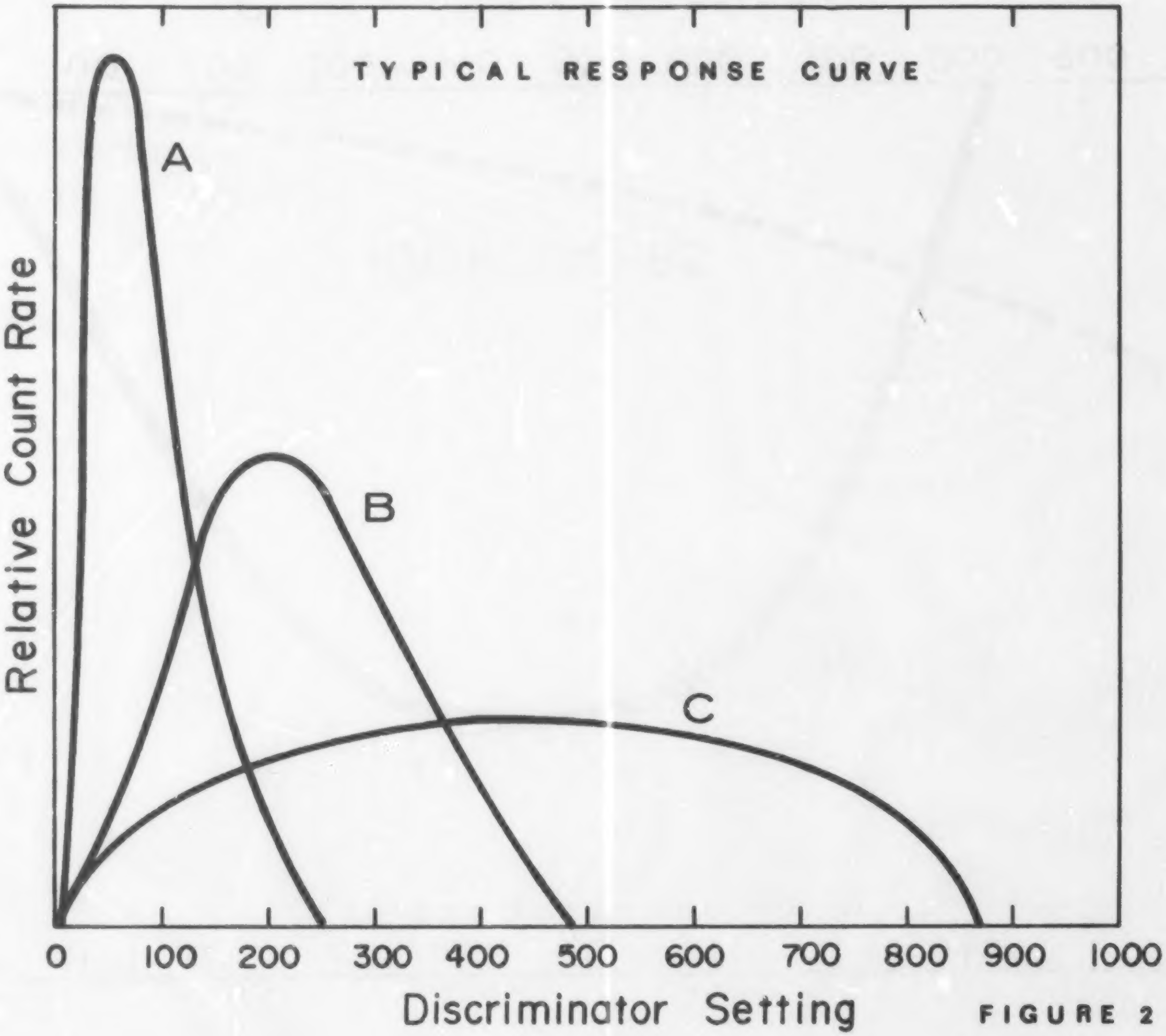

TYPICAL RESPONSE CURVE
A
B
C
Relative Count Rate
0
100
200
300
400
500
600
700
800
900
1000
Discriminator Setting

FIGURE 2

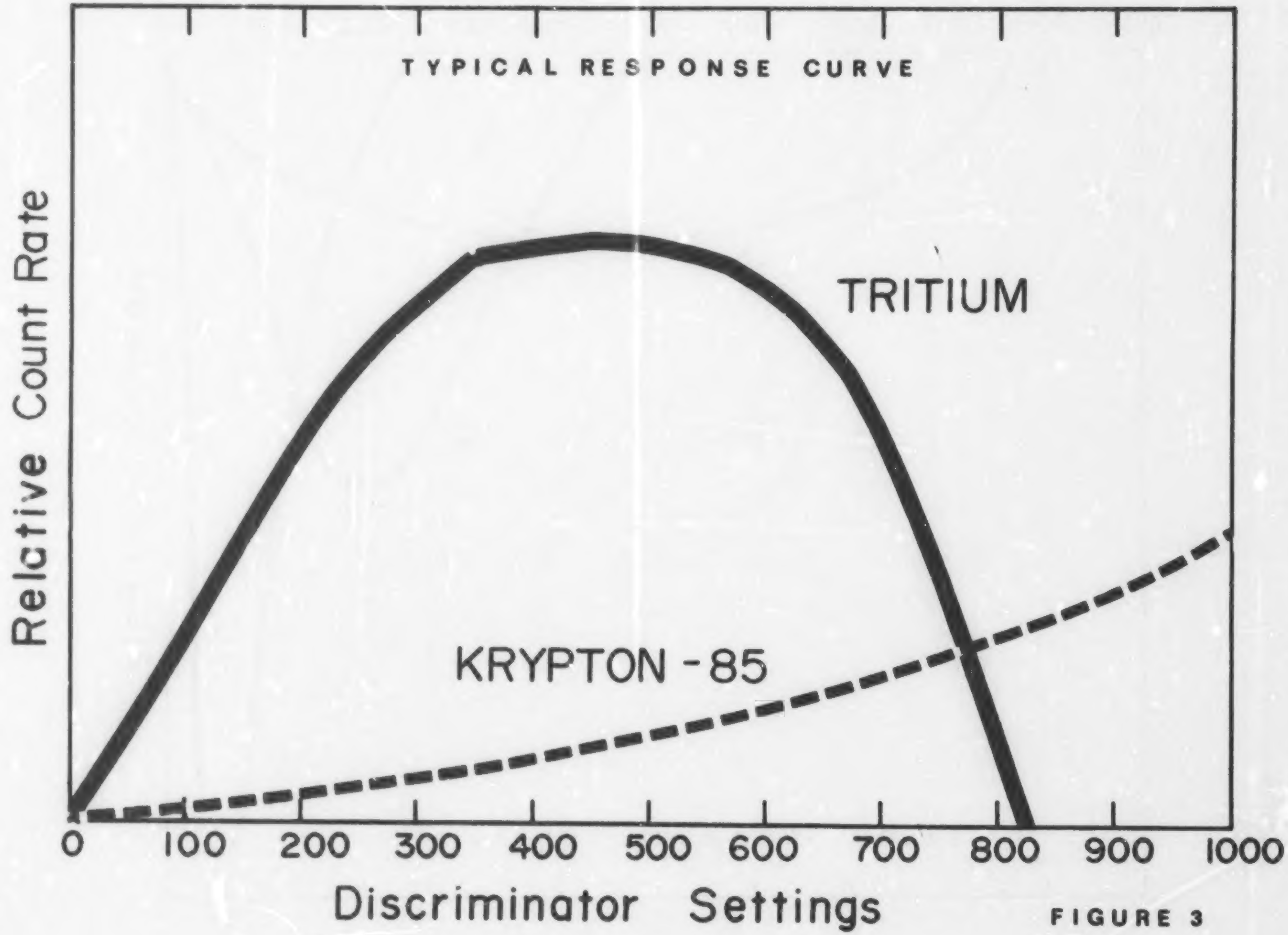

TYPICAL RESPONSE CURVE
TRITIUM
KRYPTON -85
Relative Count Rate
Discriminator Settings
0
100
200
300
400
500
600
700
800
900
1000

FIGURE 3

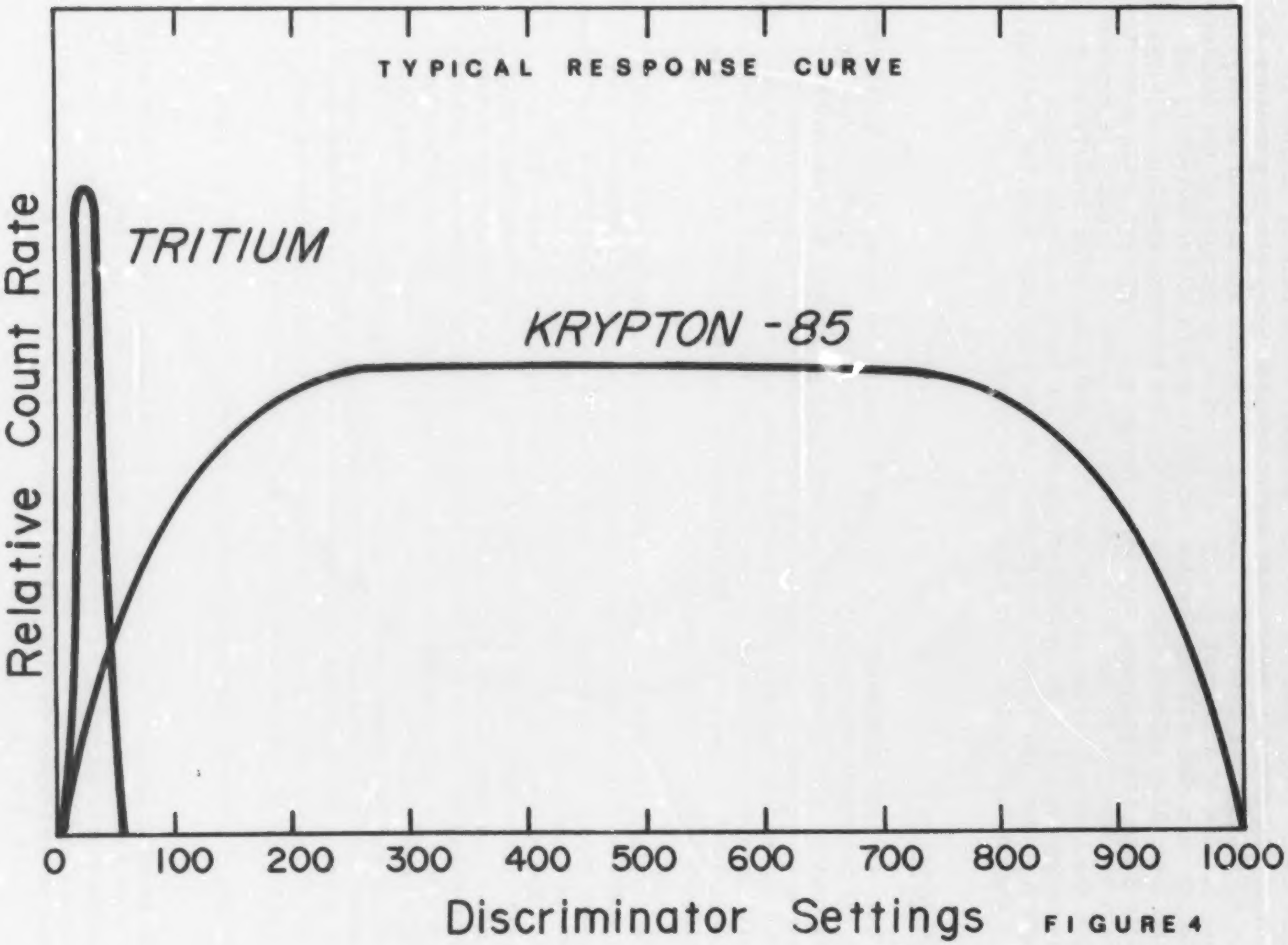

TYPICAL RESPONSE CURVE
TRITIUM
KRYPTON -85
Relative Count Rate
0
100
200
300
400
500
600
700
800
900
1000
Discriminator Settings

FIGURE 4

insinuates that, when the external standard count rate of samples is identical, then the samples are directly comparable. It is a measure as to whether the detection efficiency of tritium and krypton is varying because of different chemical and physical properties of the samples. To set this channel, the krypton sample is placed in the detector, and the lower discriminator set at 1000-00. The external standard is blown into place and the sample counted. The gain is increased until a count rate is observed in the third channel. The external standard is removed and the upper discriminator is set at 1000 and the lower discriminator is decreased until the count rate due to the krypton sample begins to appear. Then the lower discriminator is set slightly above the setting so that no counts due to krypton are recorded.

Calibration

The detection efficiency for tritium is easily measured, since tritiated water standards are readily available. Transfer to a counting vial an aliquot of a tritiated water standard equal in volume to that which is going to be used in the reaeration determinations. Fill to capacity with scintillation solution, swirl to mix, and count. The detection efficiency is calculated by the dividing of the observed count rate in the tritium channel by the disintegration per minute of tritium in the aliquot taken from the standard.

Calibrating the channel used to measure the krypton is more difficult insofar as a standard krypton source is not available. However, the gamma emissions, although only 0.4% abundant, are sufficiently energetic so as to measure if enough krypton is present. The photon energy is 514 kev which is fortunately the same as strontium-85 and which is readily available as a standardized source. To calibrate the krypton channel, a very active krypton source (10^6 disintegrations per minute) is counted in the liquid scintillation counter under the same counting protocol for the reaeration study. The beta count rate is determined. This same sample is then counted by gamma scintillation counting techniques and the gamma count rate determined. The gamma counter is calibrated using strontium-85 and after correcting for the gamma abundance of krypton-85, the beta disintegration rate is calculated. The beta scintillation efficiency is calculated by dividing the observed beta count rate by this calculated beta disintegration rate.

Figure 5 presents the gamma spectral shapes of krypton-85 and strontium-85.

If gamma scintillation counting is not available, the use of chlorine-36, a pure beta emitter whose maximum energy (714 kev) is very close to the maximum beta energy of krypton-85 (670 kev), is recommended. In this technique, however, it is assumed that the beta spectral shapes of the two radionuclides are identical or nearly identical.

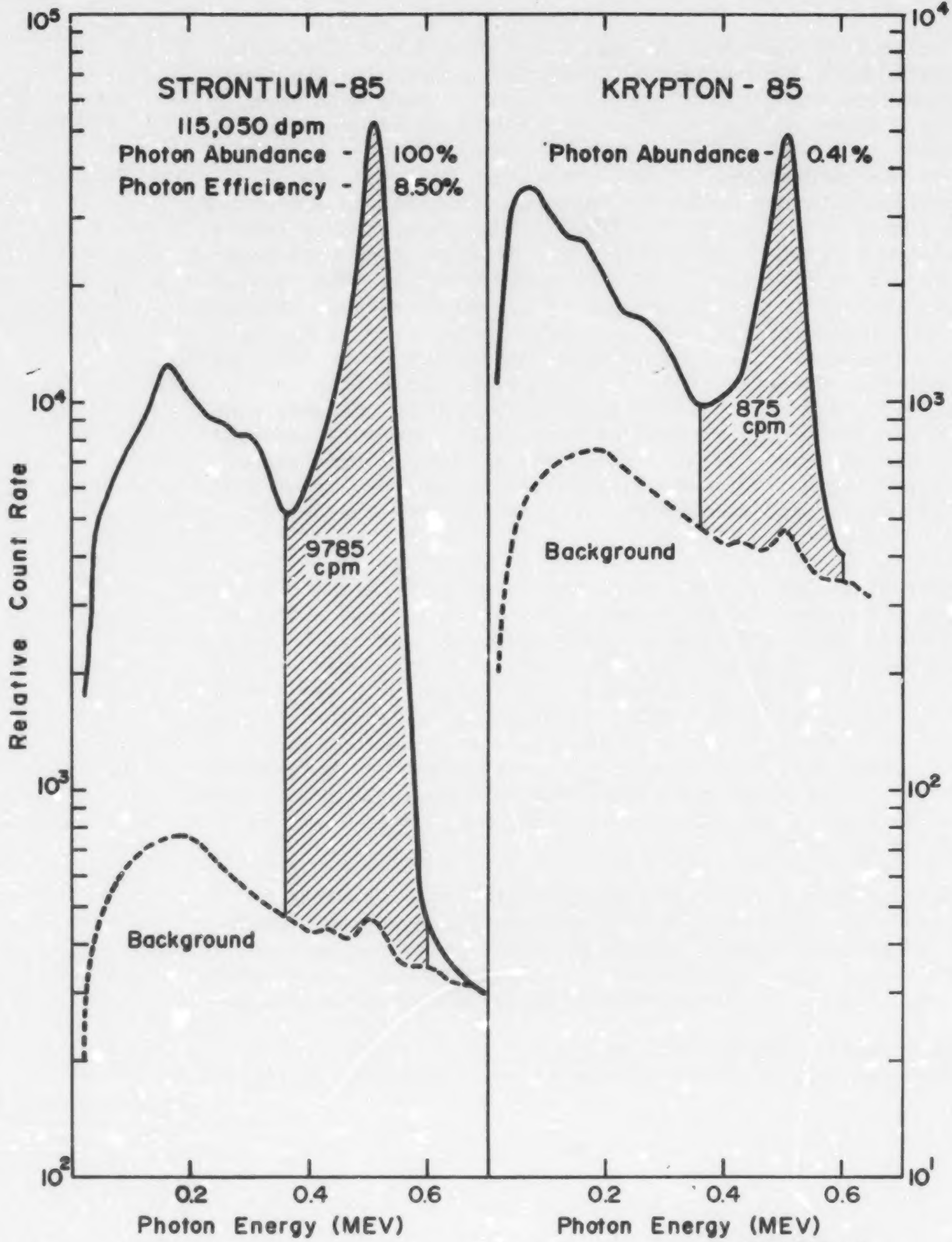

STRONTIUM-85
115,050 dpm
Photon Abundance - 100%
Photon Efficiency - 8.50%
9785 cpm
Background
KRYPTON - 85
Photon Abundance - 0.41%
875 cpm
Background
Relative Count Rate
10^5
10^4
10^3
10^2
10^4
10^3
10^2
10^1
0.2
0.4
0.6
Photon Energy (MEV)
0.2
0.4
0.6
Photon Energy (MEV)

FIGURE 5

Miscellaneous

Although not related to the topic of laboratory procedures, I wish to present some data which our laboratory determined concerning the handling of samples from the field to the laboratory. These data were obtained during a study on the James River model at Vicksburg, Mississippi. We wished to see if air transportation could be used to ship samples back to the laboratory. It was conjecture that air travel, because of possible pressure gradients between high altitude and ground, would cause a significant loss in the dissolved krypton. Forty sample bottles containing krypton-85 were prepared. Half of these were taped with plastic electrician's tape. All were gamma counted. Ten samples, five taped and five untaped, were shipped by air to Vicksburg. Another ten samples, five taped and five untaped, were transported by car to Vicksburg, while the remainder stayed in the laboratory under reasonable temperature control. When the study was completed, those that were shipped by air were returned by air and those shipped by car were also returned by car. All forty samples were recounted. The time interval between counts was 12 days. Table 1 gives the results of this experiment and our conclusions. Not stated, however, is the conclusion that it appears feasible to ship samples by air provided they are sealed with tape.

In reaeration studies, dye is used as an indicator as to when to begin sampling. Most often used is Rhodamine WT, since it is less absorbed in the stream environment rather than other dyes and thus can be used as a secondary measure of dispersion. However, this dye, when present in sufficient concentrations, does quench or shift the beta spectra of both the tritium and krypton-85. Thus, calibration is required by any of these shifts. However, it has been observed in this laboratory that uranine at these same concentrations is completely decolorized by the scintillation solution and no spectra shifts occur. If dye is used solely for the purpose of an indicator for sampling, uranine may be more advantageous to use.

One last remark concerning reaeration studies is worthy of mention. Most studies have been made using a 2 ml aliquot for counting. There is no reason, other than extreme turbidity, why 4 ml cannot be used. This, however, is the upper limit because of solubility in the scintillation solution. The use of 4 ml aliquots would do one of two things:

1. Give greater coverage to a stream, or
2. Reduce the radioactivity concentration in the dose solution.

TABLE 1

TRANSPORTATION STUDY USING KRYPTON-85

cpm gamma radiation

	Controls				Air				Car			
	Taped		Untaped		Taped		Untaped		Taped		Untaped	
	Pre	Post	Pre	Post	Pre	Post	Pre	Post	Pre	Post	Pre	Post
	3581	3398	3560	3154	3716	3474	3676	3442	3715	3446	3643	3208
	3603	3316	3546	2481	3814	3507	3643	3328	3755	3385	3672	1554
	3589	3325	3515	2055	3738	3388	3790	3351	3679	3455	3758	3337
	3597	3350	3577	3252	3743	3478	3741	2867	3673	3318	3734	3441
	3556	3397	3490	3236	3679	3560	3739	3142	3750	3307	3709	3303
	3575	3371	3491	2284								
	3561	3325	3529	3150								
	3590	3427	3502	3306								
	3538	3258	3489	3139								
	3490	3376										
Percent retention after 12 days		94		82		93		87		91		80
One standard deviation		2		14		2		6		2		22

CONCLUSIONS:

1. Based on a statistical evaluation, taped samples were more consistent and had lower overall losses than the untaped samples.
2. Within the taped samples, there was some indication that the mode of travel contributed to loss of the sample but not to a large degree.

Reaeration Capacity of the Flint South and Patuxent Rivers

E. C. Tsivoglou

The next several presentations will focus on the results obtained through application of the reaeration tracer technique in a number of streams. This first discussion outlines results for three small streams, the Flint, South and Patuxent. Subsequently, results obtained in a larger stream, the Chattahoochee, will be presented, then results from a physical model of a large tidal stream, and, finally, field results observed in a small tidal stream in Oregon.

As outlined earlier, the principal purpose of these field studies of reaeration capacity has been to observe reaeration rates in streams that include a wide range of hydraulic features, so as to facilitate study of the basic relationships between reaeration and stream hydraulic properties. This should then permit analysis of the accuracy and dependability of currently available predictive models for reaeration, and, possibly, some improvement of these models. To the extent indicated by the observed results, new predictive models may also be developed and tested. As a secondary, but practical, purpose, the field studies have been designed to provide useful information for real streams having real pollution problems.

In reviewing the next few figures, which show observed results for three small streams, it should be recalled that reaeration, or gas transfer, is treated as a first-order process, as usual. Therefore, if the observed krypton:tritium ratios are plotted vs. time of flow on semilog scales, the slope of the line between any two sampling stations represents the reaeration coefficient, K_2, for the krypton tracer, to the common log base. Therefore, the steeper the line, the higher the reaeration capacity of the stream section.

Patuxent River

Figure 1 is a map of the section of the Patuxent River that has been studied. The Patuxent is a coastal plains stream in Maryland that flows through an important wildlife refuge. It receives a highly treated (activated sludge process) municipal waste at the head of the study section, just above the wildlife refuge, and the study section was the eight-mile reach below the waste treatment plant effluent. No other wastes enter the stream in the study section.

The Patuxent is characterized by gentle shallow riffles and slow pools, with no waterfalls, rapids or white water. The reaeration studies were conducted at critical low flows and summer temperatures.

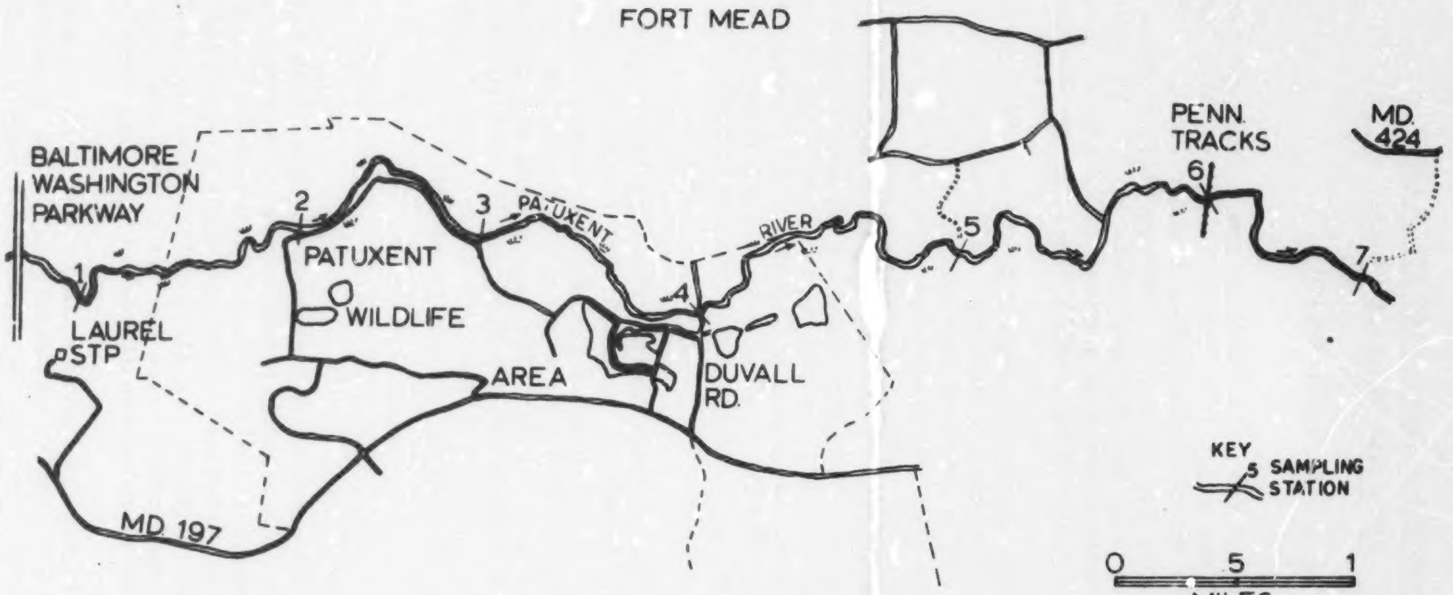


FIGURE 1
PATUXENT RIVER STUDY LOCALE

At the prevailing stream flows of eight or 10 cfs at the head of the study section, depths of flow through the study reach ranged from a few inches to two or three feet in some pools. Tributary branches increased the flow to about 24 cfs at the lower end of the study section, and the time of flow for the entire reach studied was about 30 hours.

The Patuxent River studies were conducted jointly with the Maryland Department of Water Resources.

Figure 2 shows typical study results for the first four miles below the Laurel, Md., waste treatment plant (about 14 hours time of flow). The results have been plotted on semilog paper, so that the slopes of the lines directly represent the reaeration rate coefficients. Results for two separate tracer releases are shown, representing flows at the head of the study section of 10 cfs and 17 cfs, respectively.

Referring to Figure 2, several observations are noteworthy. First, the observed reaeration rate coefficients are quite consistent, from one reach to the next, indicating an absence of unusual or remarkably different hydraulic features. The relative slopes for any one section are also highly consistent for both tracer dumps, and indicate that, even though the flow for Dump D was nearly double that for Dump B, the observed reaeration rate coefficients were only slightly higher. The range of K_2 values was quite small (0.09 to 0.15 per hour) for both dumps. In fact, for this coastal plains stream, no important error would be made by using a single average value of K_2 for the whole reach, Stations 1 to 4. The single representative values of K_2 would be 0.12 per hour at about 10 cfs and 0.13 per hour at 17 cfs, at 25°C.

The reproducibility of results was excellent in these studies.

South River

Figure 3 is a map of the South River tracer study section. The South River originates in southwest Atlanta, Ga. The study reach was about 18 miles long, beginning at the South River Sewage Treatment Plant in Atlanta. With a small number of notable exceptions, the channel is relatively uniform, with typical depths of one to two feet and typical widths of 30 to 40 feet. The bottom is usually sandy.

The South River is presently heavily polluted. It receives partially treated and some untreated sewage and industrial waste from four waste treatment plants in the 18-mile study section. These include the South River STP, the Intrenchment Creek STP, the Shoal Creek STP and the Snapfinger Creek STP. All of these plants are overloaded. Typical flows in the South River range from about

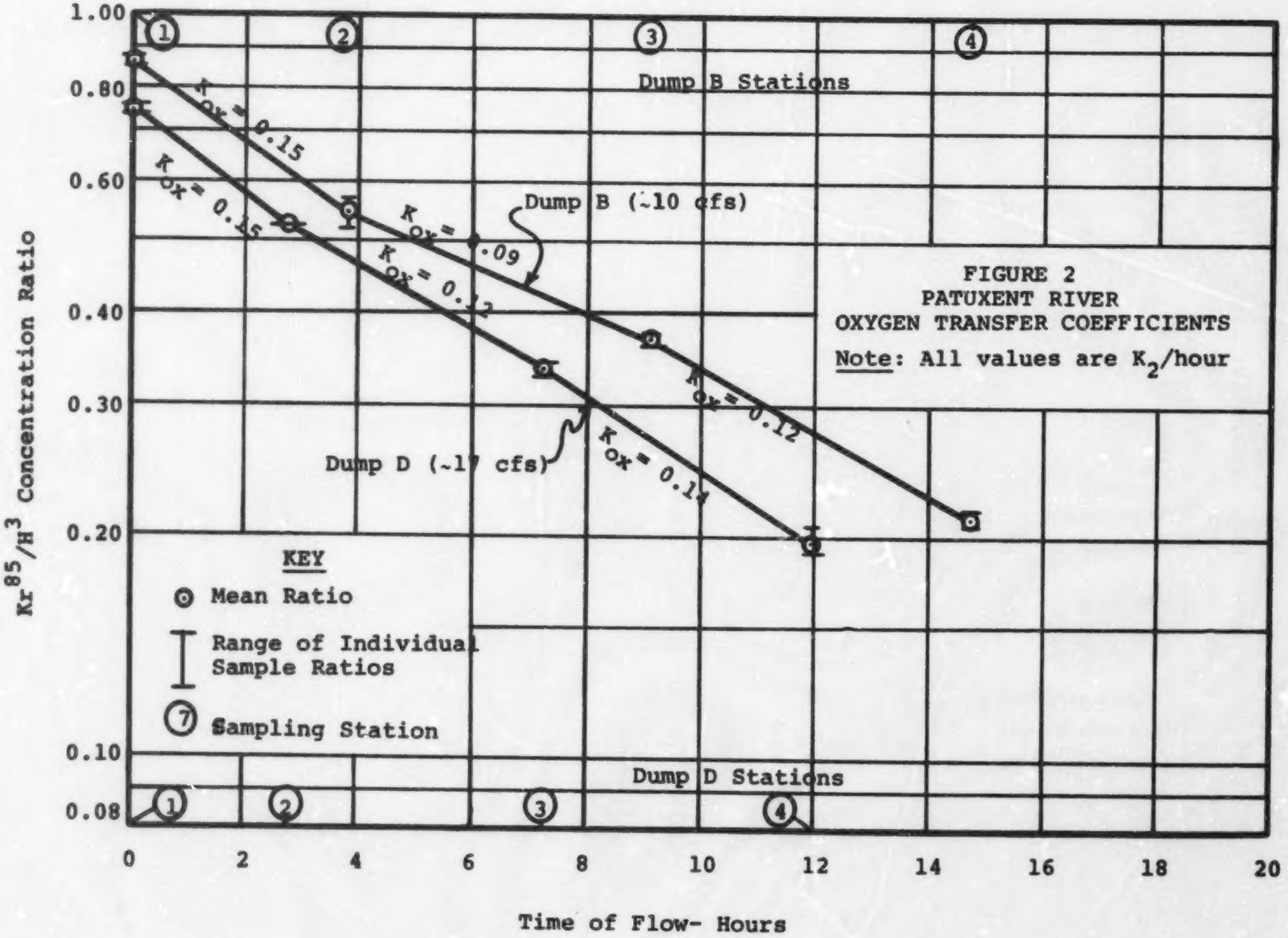


FIGURE 2
PATUXENT RIVER
OXYGEN TRANSFER COEFFICIENTS

Note: All values are K_2/hour

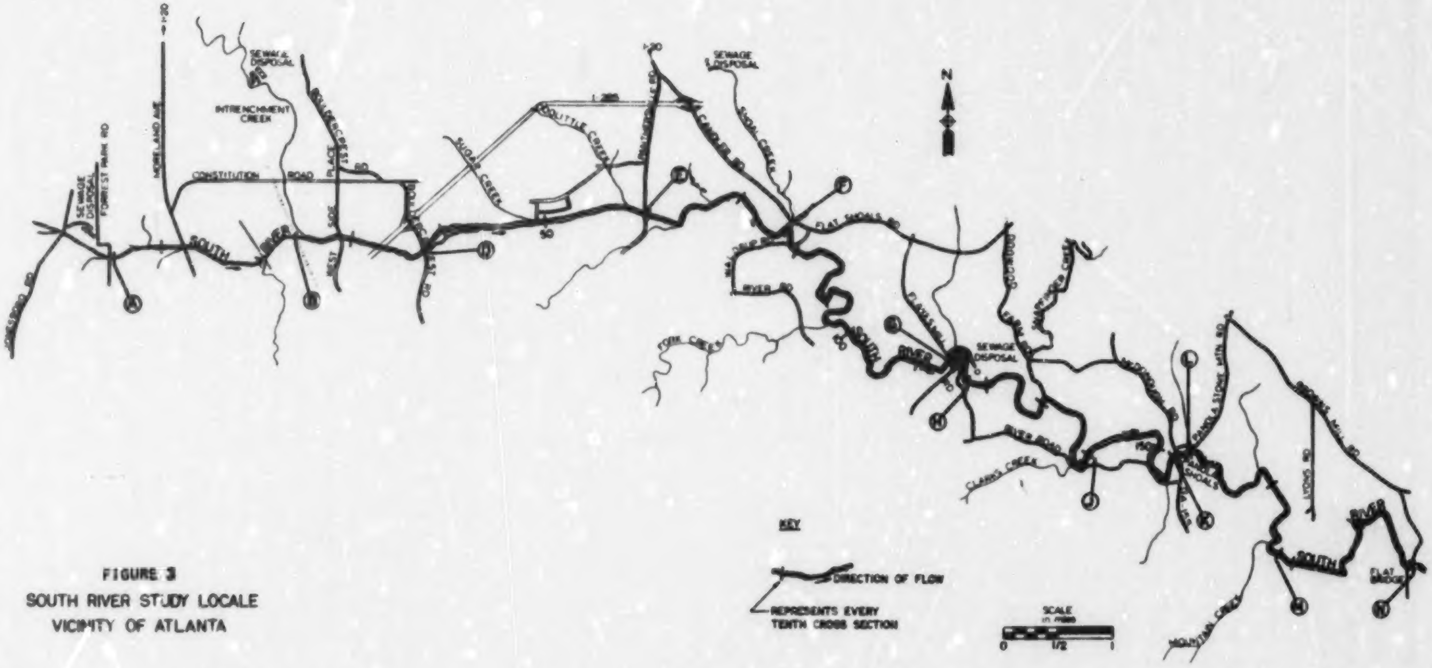


FIGURE 3
SOUTH RIVER STUDY LOCALE
VICINITY OF ATLANTA

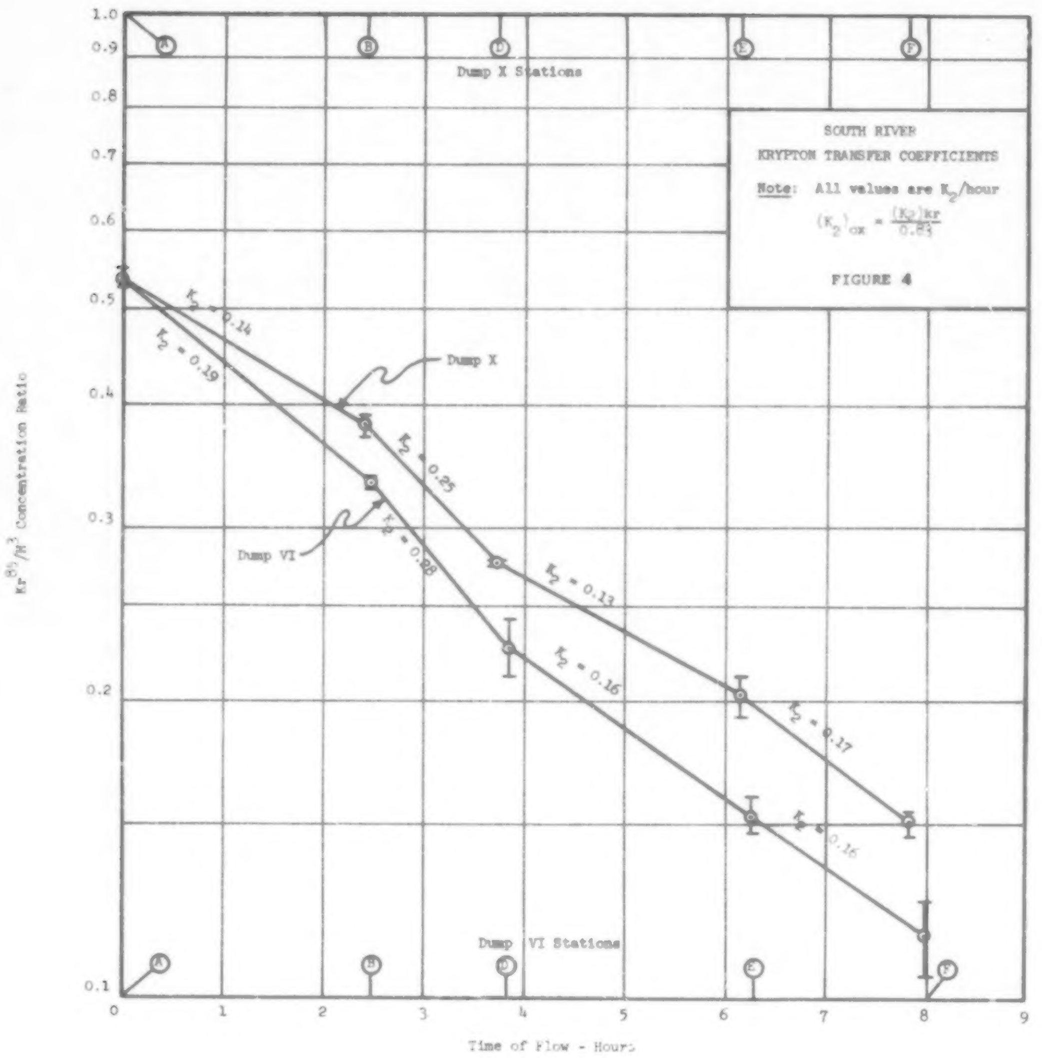
50 cfs at the upper end of the study section to perhaps 200 cfs at the lower end. Foam is often observable in the stream in the summertime, especially below the Snapfinger Creek STP and in the pool just below Panola Shoals.

A number of tracer releases were made in the South River, and the next three figures illustrate the results obtained. Figure 4 shows the results of two such releases at the head end of the study section. As indicated by the times of flow for the two dumps, stream flow was not greatly different in the two studies. The observed K_2 values ranged from 0.13 to 0.28 per hour in the different reaches. For any one reach, the reproducibility of results from one dump to the next was very good. The reach BD exhibited some difference from the others, having a significantly higher value of K_2 in both dumps, and does contain a short shallower rapids section.

Figure 5 shows the results of five separate tracer dumps, covering the middle seven to eight miles of the 18-mile study section, from above Shoal Creek through the large pool just above Panola Shoals. As may be seen, the reproducibility of results from one dump to the next was generally very good, especially in the reaches FG and HJ. A wide range of K_2 values was observed, from 0.05 to 0.74 per hour for krypton transfer. Results for the section GH were more variable, the K_2 values ranging from 0.39 to 0.74 per hour for krypton. Section GH contains a violent rapids section 200 to 300 feet long, with much white water, and is just below the entry of the effluent from the Shoal Creek STP. The effect of the rapids section is quite evident in terms of the high gas transfer coefficients. Results for the reach JK, which includes the large pool above Panola Shoals, are also somewhat more variable (K_2 ranging from 0.05 to 0.11 per hour for krypton) for reasons outlined below.

The Panola Shoals section was studied more intensively because of its unusual hydraulic nature, and Figure 6 shows the results of those studies. All of the K_2 values shown represent krypton transfer, but can be converted to K_2's for oxygen transfer by dividing them by the basic conversion factor of 0.83. At first glance, these results appear to be more variable, both for the pool and for the shoals themselves. Note, however, that the observed results for the entire reach are highly consistent and reproducible, ranging only from 0.17 to 0.19 per hour for the four tracer dumps. Further analysis of these results has resulted in a better understanding of the reasons for the apparent variability of results in the pool and over the shoals, and has indicated a possible source of difficulty in field operations.

Flow through the long deep pool just above Panola Shoals is decidedly not homogeneous or uniform. Thus, flow from above may at times remain near the pool surface, rather than mixing throughout the whole depth of the pool. Under other circumstances, the reverse may well happen. Thus, the location of the sampling point (the depth of sampling) just above the shoals could have been less representative



SOUTH RIVER
KRYPTON TRANSFER COEFFICIENTS

Note: All values are K_2/hour

$$(K_2)_{ox} = \frac{(K_2)_{kr}}{0.83}$$

FIGURE 4

R

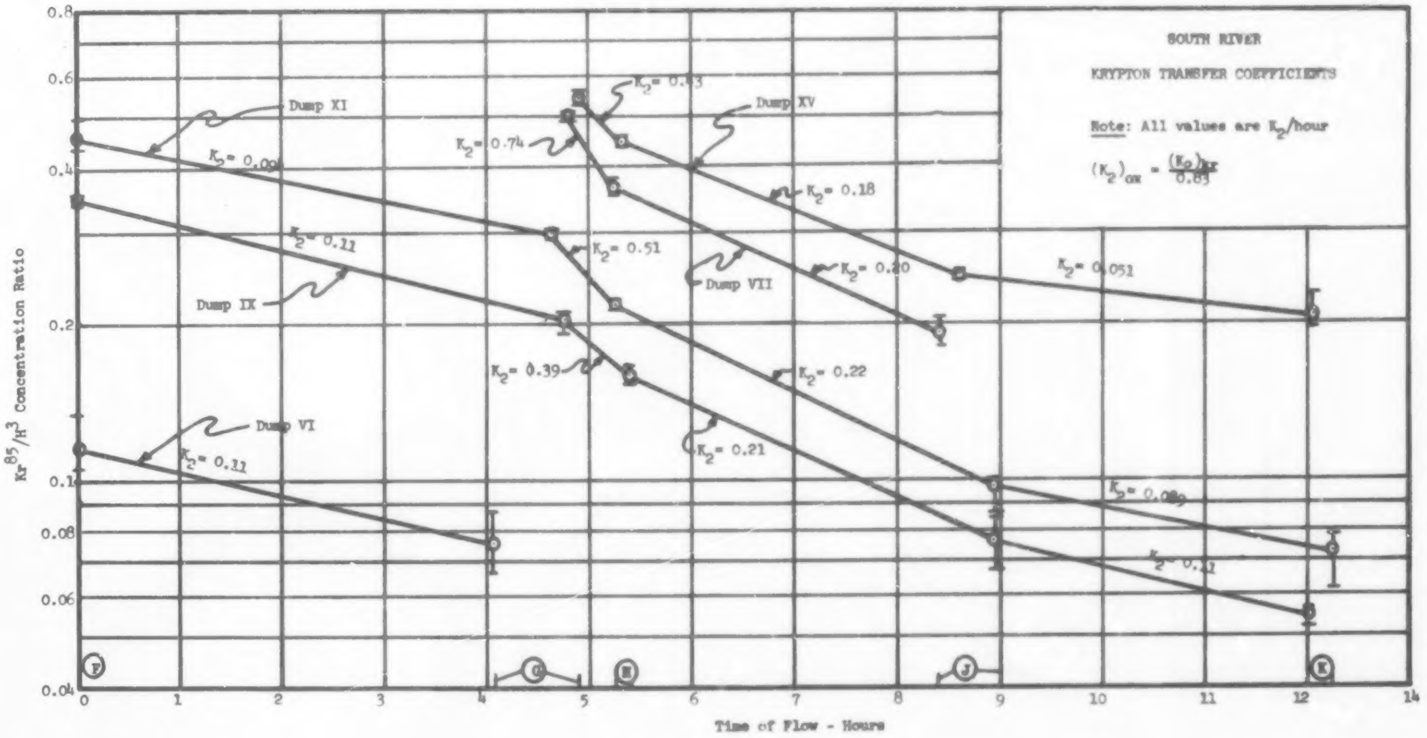

SOUTH RIVER
KRYPTON TRANSFER COEFFICIENTS
Note: All values are K_2/hour
$(K_2)_{ox} = \frac{(K_2)_{kr}}{0.83}$
Dump XI
K_2= 0.09
Dump IX
K_2= 0.11
Dump VI
K_2= 0.11
K_2= 0.74
K_2= 0.43
Dump XV
K_2= 0.18
K_2= 0.51
Dump VII
K_2= 0.20
K_2= 0.051
K_2= 0.39
K_2= 0.22
K_2= 0.21
K_2= 0.089
K_2= 0.11
F
G
H
J
K
Kr^{85}/H^3 Concentration Ratio
0.8
0.6
0.4
0.2
0.1
0.08
0.06
0.04
0
1
2
3
4
5
6
7
8
9
10
11
12
14
Time of Flow - Hours

Figure 5

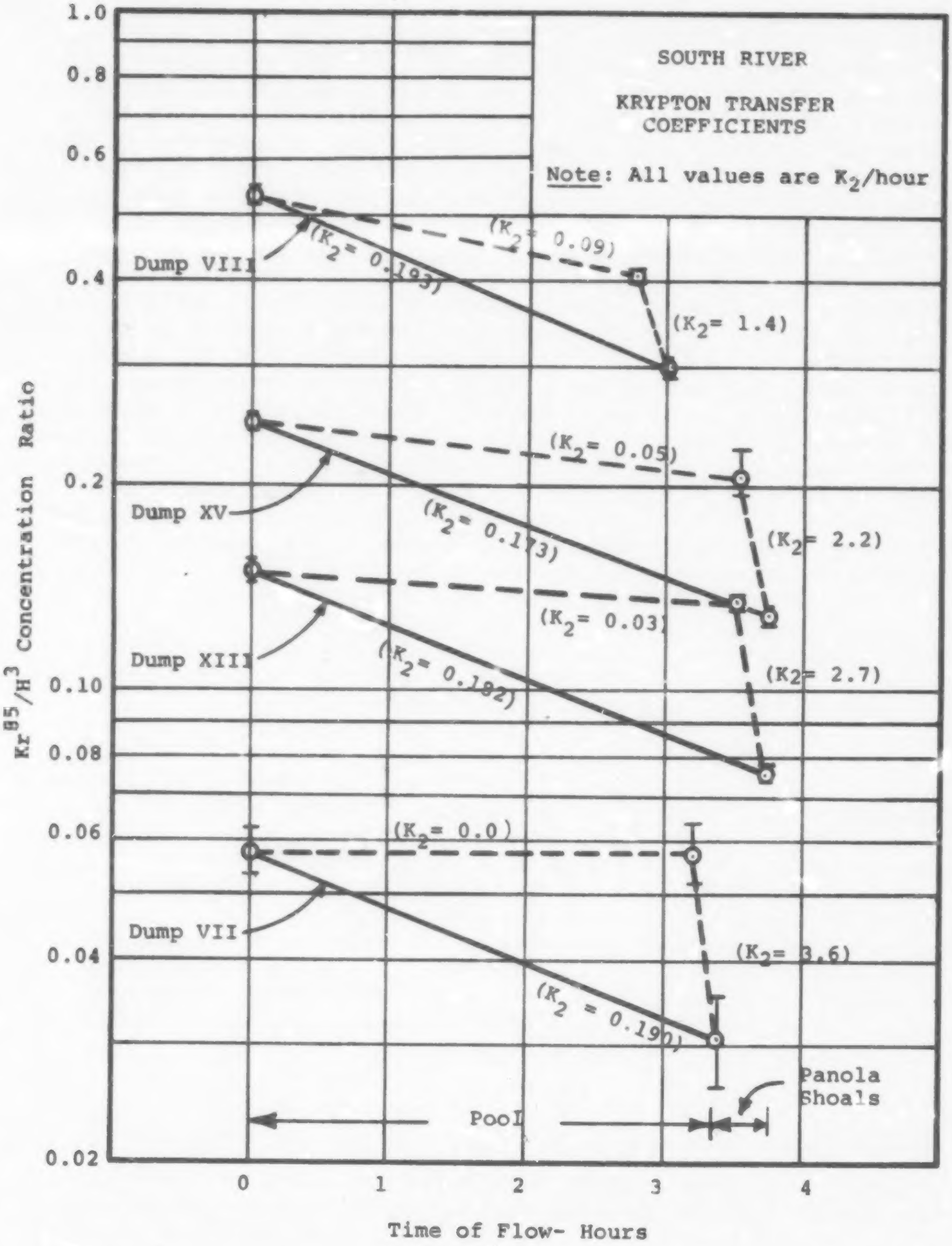

SOUTH RIVER
KRYPTON TRANSFER COEFFICIENTS
Note: All values are K2/hour
1.0
0.8
0.6
0.4
0.2
0.10
0.08
0.06
0.04
0.02
Kr85/H3 Concentration Ratio
Dump VIII
(K2= 0.09)
(K2= 0.193)
(K2= 1.4)
Dump XV
(K2= 0.05)
(K2= 0.173)
(K2= 2.2)
Dump XIII
(K2= 0.03)
(K2= 0.182)
(K2= 2.7)
Dump VII
(K2= 0.0)
(K2= 3.6)
(K2 = 0.190)
Pool
Panola Shoals
0
1
2
3
4
Time of Flow- Hours

FIGURE 6

than necessary in one study or another - although the results obtained would be accurate for that sampling depth, they might not adequately reflect the mixing action of the whole pool volume. In addition, the time of flow over the shoals themselves was very short - about 13 minutes between the two sampling stations, and this introduces the possibility of error in terms of the observed time of flow over such short periods. Three of the four dumps shown in Figure 6 were made at the Station immediately above the pool, and under such circumstances it is not impossible that the sampling point immediately above the shoals was not always in the best location across the width of the pool: time of flow is determined by the time lapse between peak dye concentrations at successive sampling points, and the pool just above the shoals is very wide; if the upper sampling point were not located to catch the exact peak dye concentration, but somewhat to one side, an error of two or three minutes in the observed time of flow could well results. Normally, the time of flow between sampling stations is two or three hours, and such an error would be entirely negligible. But for short times, such as the 10-15 minutes over Panola Shoals, an error of 2 or 3 minutes would result in a substantial error in the calculated value of K_2. This may well have happened in the Panola Shoals reach because of the combined circumstances of dosing not far upstream, nonuniform mixing in the pool and the width of the sampling station just above the shoals. The excellent consistency of results for the whole reach, vs. the apparent variability within the pool and over the shoals taken separately, clearly indicates that the exact location of the sampling point just above the shoals was crucial for best results.

For the pool above Panola Shoals the average observed K_2 of 0.04 per hour for krypton (0.05 per hour for oxygen) is regarded as close. Similarly, the average K_2 of 2.5 per hour for krypton across the shoals (K_{ox} = 3.0 per hour) is taken to be satisfactory. Thus, across Panola Shoals, in a period of about 13 minutes, about 40 per cent of the available krypton-85 tracer gas was lost to the atmosphere. In terms of reaeration, this means that about 50 per cent of the DO deficit was overcome across these shoals. This is, of course, a very high rate of reaeration, and was shown to be quite correct by independent DO sampling at the time of each tracer dump. These results clearly demonstrate the great effectiveness of sudden water surface elevation changes in bringing about stream reaeration.

Flint River

Figure 7 is a general map of the Flint River study section. The Flint also rises in southwest Atlanta and flows generally southward. The study section was about 10 miles long, beginning below the Flint River Sewage Treatment Plant, just south of the Atlanta airport. The stream was heavily polluted during all of our studies, as the Flint River STP is also badly overloaded, and foam was commonly observed below rapids and falls. Typical flows during these studies

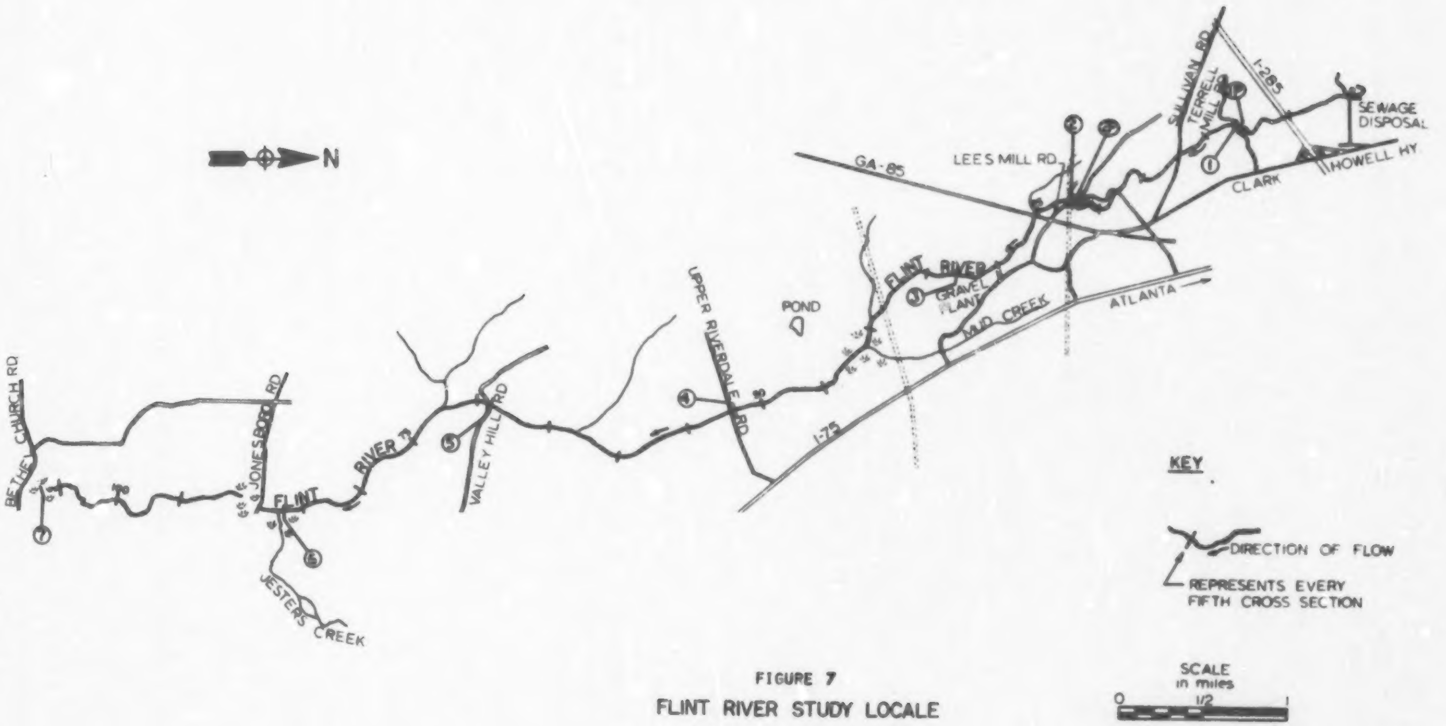


FIGURE 7
FLINT RIVER STUDY LOCALE
VICINITY OF ATLANTA

ranged from 5 cfs just below the STP to 30 cfs or so at the lower end of the study reach, and depths of flow ranged from one or two inches to two feet or more in pools.

The study section on the Flint contains a number of unusual hydraulic features, and there is no "typical" section. The stream is made up of alternating riffles and pools. At the upper end there is a violent rapids reach perhaps 50 yards in length and containing several small hydraulic jumps. Below, there are two old mill ponds, each followed in turn by a waterfall 12 feet or so high. Farther downstream, the Flint passes through a marsh in multiple small channels. The following figures and charts illustrate the effects of some of these features on the reaeration capacity.

Figure 8 is a semilog plot of the results observed in two separate tracer releases just below the Flint River STP. Dump No. III was at a flow of 10 cfs, while Dump XIV was at a flow of only 5.5 cfs, the lowest encountered. Referring to Figure 8, it is clear that the slopes of the lines shown, and hence the K_2 values, were not remarkably affected by the difference in stream flow, even though the time of flow was substantially different. Reach 01P contained a violent rapids section, R1R3, with small hydraulic jumps, followed by a long, shallow pool, R31P; the next reach, 1P1, was a waterfall about 14 feet high, with shallow flow over smooth rocks - the time of flow from Station 1P to 1 was 5 or 6 minutes; the reach 1 to 2P consisted of alternating riffles and small pools until the stream entered a larger long pool above Station 2P (with depths up to 3 feet and muddy bottom); the reach 2P2 was a second waterfall, about 12 feet high, with a short deep pool immediately under the fall - time of flow from Station 2P to Station 2 was of the order of 25 minutes or so, largely in the pool under the falls.

The pattern of gas transfer and reaeration is remarkably consistent, and the results highly reproducible, as shown in Figure 8. They show clearly the effect of features such as rapids, waterfalls and pools in determining gas transfer and reaeration, with K_2 values for krypton as high as 12 per hour at the waterfalls and as low as 0.015 per hour in the pools. For Dump XIV, the results also demonstrate that in streams of this type, having distinct hydraulic features such as those shown, most of the real work of gas transfer takes place in relatively short reaches and times: the rapids section R1R3 is where most of the work of gas transfer takes place in the long reach 01P, and very little such work is accomplished in the pool between stations R3 and 1P. In contrast, the time of flow through the rapids was about 7 minutes, and through the pool almost 3 hours.

Considered another way, in Dump XIV, a total of 44 per cent of the tracer gas was lost during the 4.7 hours time of flow between Stations 0 and 1P; of this total lost, about 56 per cent took place in the 1.7 hours between Stations 0 and R1, about 38 per cent took place in the 7 minutes between R1 and R3, and only about 6 per cent took place in

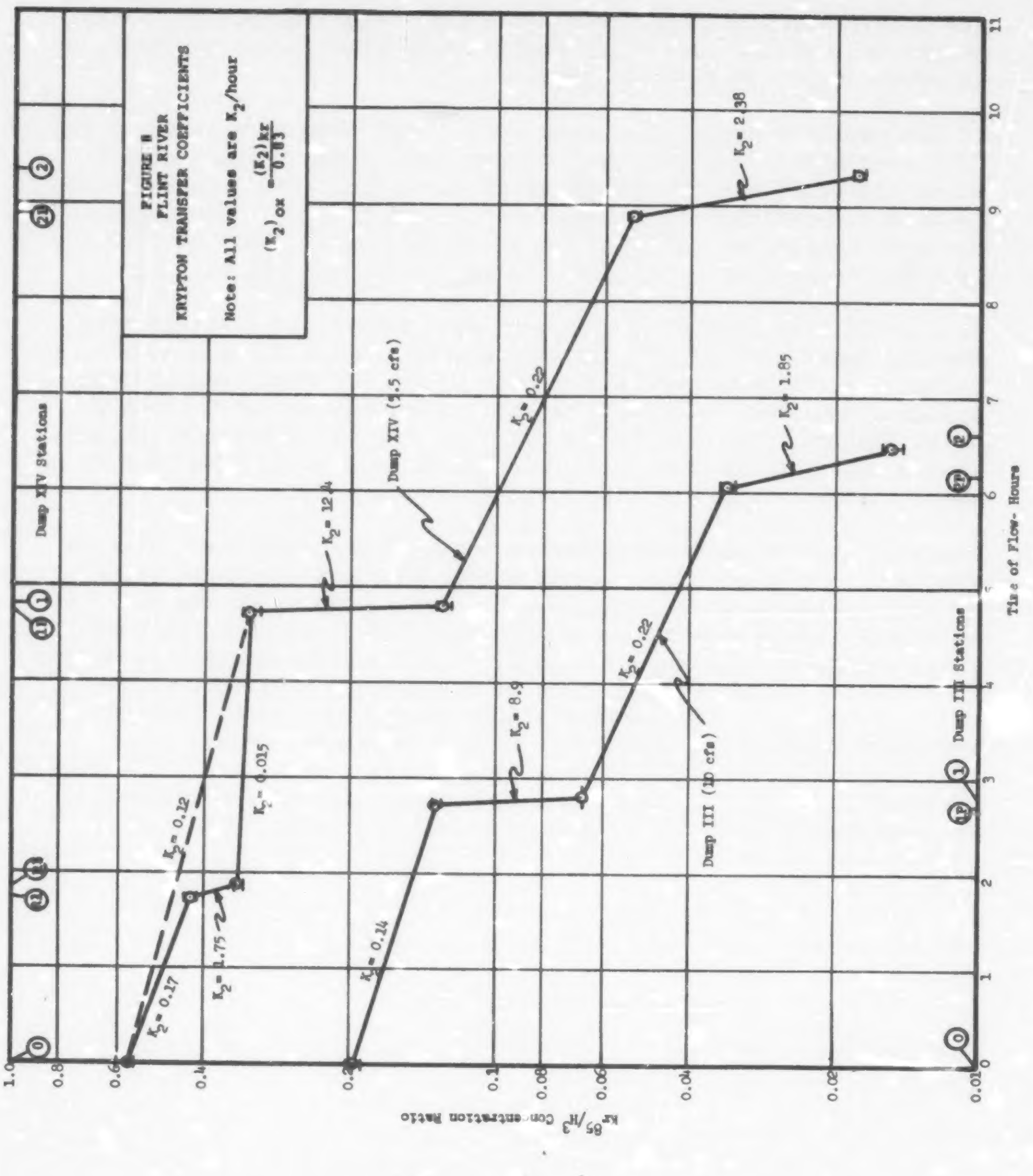


FIGURE 8
FLINT RIVER
KRYPTON TRANSFER COEFFICIENTS

Note: All values are K_2/hour $(K_2)_{ox} = \frac{(K_2)kr}{0.83}$

the 3 hours time of flow through the pool below Station R3. Thus, 94 per cent of the work of reaeration took place above the pool, or in about 38 per cent of the time.

At both waterfalls, about 55 per cent of the tracer gas present was lost over the falls, and about 65 per cent of the oxygen deficit above the falls was overcome through the falls. These results were confirmed by independent DO analyses for each study. The reported K_2 values differ sharply because of the different times of flow - because of the deep pool located under the second falls.

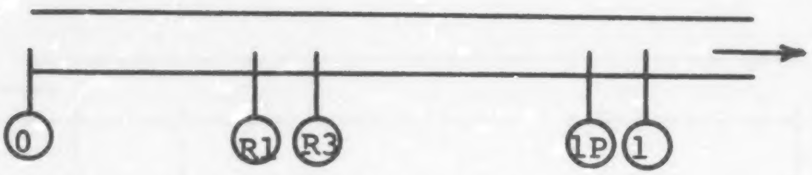
In Dump III, at 10 cfs, 92 per cent of the tracer gas was lost in the 6.5 hours time of flow between stations 0 and 2; in Dump XIV, at 5.5 cfs, the loss was 97 per cent in the 9.3 hours required to traverse the same distance. In both cases, any oxygen deficit per se at Station 0 was completely overcome, and had there been no pollution load also present, a DO deficit of zero would have been found at Station 2.

Figures 9 and 10 tabulate a few of the foregoing results, for convenience. Clearly, in a natural stream there is no single value of K_2 that really prevails over long distances, except in the sense of an average result, as so clearly shown for the reach 01P in Dump XIV. Thus, although gas transfer is a first-order process under condtions of uniformly steady mixing, in a natural stream many such first-order processes occur within a single reach, due to the lack of hydraulic uniformity, and observed "K_2" represents an average result rather than a single reaction. In many ways, then, the actual observed per cent gas loss seems a more significant representation of gas transfer and reaeration than a calculated value of K_2.

FIGURE 9

FLINT RIVER

(Summer, 1969)

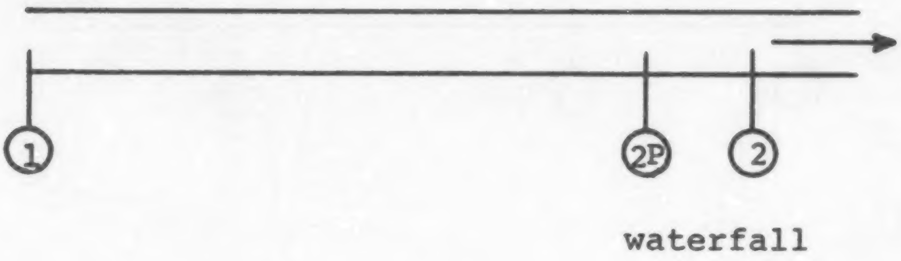


Reach	Dump	Percent Gas Loss	(K_2)Kr/hr
01	I	(49.8)	(0.224)
	III	65.9	0.366
	XIV	77.7	0.312
01P	III	31.8	0.141
	XIV	43.5	0.121
R1R3	XIV	20.5	1.75
1P1	III	50.1	8.9
	XIV	60.5	12.4

FIGURE 10

FLINT RIVER

(Summer, 1969)



Reach	Dump	Percent Gas Loss	(K_2)Kr/hr
12	I	(75.7)	(0.352)
	II	75.0	0.401
	III	77.1	0.402
	XIV	85.1	0.434
12P	II	49.5	0.220
	III	50.3	0.215
	XIV	58.9	0.219
2P2	II	50.4	2.01
	III	53.9	1.85
	XIV	65.8	2.38

OXYGEN BALANCE OF THE SOUTH RIVER

A. G. Herndon

During the last three years the Federal Water Quality Administration, in cooperation with the Georgia Water Quality Control Board, has conducted studies on the South River to determine pollution loads and assimilation characteristics. These studies, classical in nature, included measurements of dissolved oxygen, BOD, ammonia, nitrates, and other parameters. One study consisted of daily sampling throughout the entire reach of the river over a period of three weeks. The reaeration coefficients developed by Dr. Tsivoglou, along with data from the previous studies, were used to develop a more complete evaluation of the self purification characteristics of the South River.

The South River is very polluted, even though all wastes discharged to it receive from 85 to 90% BOD removal. Pertinent data on the sewage treatment plants located in the study area are as follows:

Plant	Flow, cfs	BOD (Ultimate) lbs/day
South River	22	4,300
Entrenchment Creek	21	6,600
Shoal Creek	4	1,300
Snapfinger Creek	5	1,500

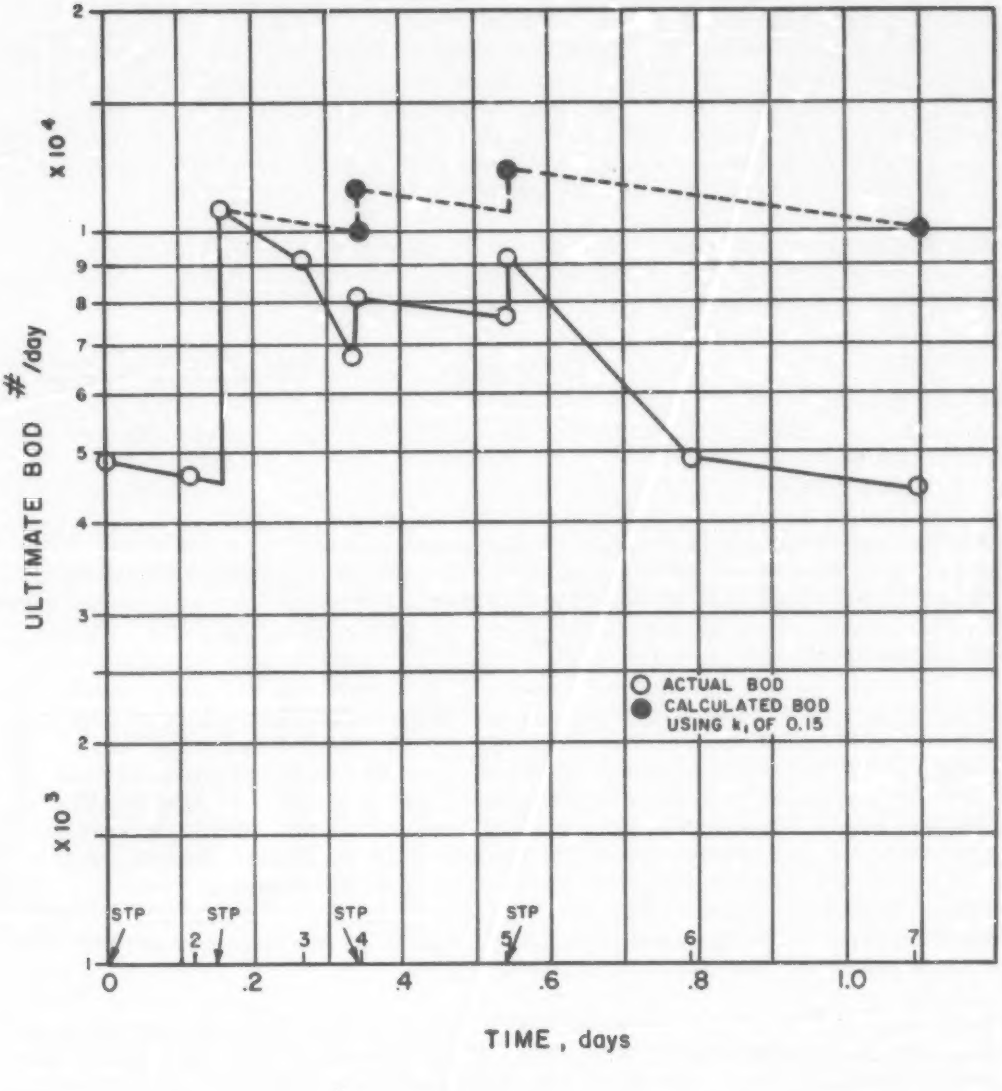
During the study in 1967, all of the plants discharged effluents with high BOD's, due to industrial wastes and excess loadings. Since the flow in the upper end of the South River is only 13 cfs and it receives 13,700 pounds of BOD_U over a distance of about 15 miles, this river has a heavy pollution load.

Data from waste loadings within the river and discharges from plants were used to plot the stream loadings as shown in Figure 1. This chart shows the ultimate BOD in pounds per day plotted against the time of travel at low flow conditions. The location of the four sewage treatment plants (STP) and the water quality sampling stations are shown on the chart. Station Number 6 is at Panola Shoals, which is at 0.8 day travel time, or about 15 miles downstream, from the first treatment plant. The solid line of the chart shows the actual measured BOD loadings, which began at 4,800 pounds per day at the discharge point of the first treatment plant.

Long-term BOD tests on river water samples showed that the deoxygenation rate (k_1) was 0.15 (base 10). When this rate was applied to the waste loadings in the stream, the data shown on Figure 1 as a broken line were calculated. This shows that about 72% of the applied BOD would remain at the lower end of the study reach. However, as shown by the solid line, only about 32% of the BOD actually remains at the lower end of the study reach.

Figure 1

SOUTH RIVER WASTE ASSIMILATION



Using the above information and the reaeration coefficients (k_2) that were measured by Dr. Tsivoglou, additional k_1 and k_2 rates were calculated, as shown in Table 1. The river was divided into relatively short reaches, which were determined by the sampling stations, numbered 1 through 6. No calculations were made on the reach between station 6 and 7 because reaeration data were not available from Dr. Tsivoglou's study.

The measured deoxygenation coefficients (k_1), or BOD reduction rates, were calculated as the slopes of the solid lines plotted in Figure 1. The reaeration coefficients (k_2) from Dr. Tsivoglou's data were converted to 17°C and per-day basis. Since his study reaches covered somewhat different distances, some of his data were averaged to obtain these k_2 values. For example, he measured k_2 in three reaches between stations 5 and 6. The temperature conversion made all data consistent with the field studies. It should be noted that the measured k_1 values varied widely, indicating a very broad range of deoxygenation or BOD reduction rates in the South River. These conditions are somewhat apparent upon observation of the stream where some sections have wide, sandy bottoms and other sections have rocky, turbulent areas.

The Streeter-Phelps equation was used to calculate k_2 by utilizing the measured k_1 from Figure 1 and dissolved oxygen data from field surveys. A comparison of the measured and calculated k_2 indicate fair agreement at some sections, but poor agreement at the section between stations number 3 and 5. The k_2 values, obtained by direct measurement, were then used to calculate corresponding k_1 values. A comparison of the measured and calculated k_1 values shows that there is considerable discrepancy in individual reaches; however, almost all the values are much higher than normally observed in larger streams.

The high deoxygenation rates (k_1) indicate that this shallow, turbulent stream is exhibiting a phenomenon referred to by some workers as biological extraction, or perhaps some other terminology. The measured k_1 rates were very high in three sections, where the values were 0.62, 1.72, 1.09. If these k_1 rates were true deoxygenation rates, then an equivalent amount of oxygen to satisfy the BOD demand would be required to maintain the dissolved oxygen level in the stream. This would then require very high reaeration rates, as was calculated in the sections between stations 3 and 4 and stations 5 and 6. It appears that in some sections of this river the BOD is being removed without requiring an equivalent oxygen demand; or else, additional oxygen is being supplied, other than that of normal reaeration. The general appearance of the stream would indicate that the rocky areas with biological growths would have a higher BOD extraction rate than those areas which have sandy, shallow bottoms.

A more detailed study of the section between stations 5 and 6 may reveal some new insight on the assimilation characteristics. In Figure 1, this section is shown to have a steep slope of BOD reduction; however, a closer examination of shorter reaches may reveal

TABLE 1

South River
Deoxygenation and Reaeration Coefficients
(17°C, Base 10, per day)

Station	Measured		Calculated	
No.	k_1 (chart)	k_2 (Tsivoglou)	k_1*	k_2**
1				
	.12	1.80	.15	1.64
2				
	.62	1.94	.83	1.23
3				
	1.72	1.62	.45	4.85
4				
	.14	1.05	.64	.17
5				
	1.09	1.77	.66	2.37
6				

*Using k_2 by Tsivoglou

**Using k_1 from chart (Figure 1)

much steeper slopes in the rocky, turbulent areas. Dr. Tsivoglou's tracer studies showed reaeration rates of 5.6, 2.2, and 0.6 per day in the three subsections of this reach. Correspondingly high and low BOD reduction rates probably occur in these same subsections; however, data were available only on the average reduction as shown in Figure 1 between stations 5 and 6.

The above data are presented, not to show that all of the answers are known, but to show that there is a wide variability in the factors affecting assimilation when measured by different methods. Also, the data point out the need for additional studies in various areas. Although the original field studies on water quality indicated no significant nitrogenous oxygen demand in the study area, there was a major demand downstream of the study area. Additional work is needed to determine the effects of algae on dissolved oxygen production, since nothing was done on this in previous studies. The biological growths on the rocks in some areas probably contribute significantly to the BOD reduction and this should be explored in short reaches of the river.

In conclusion, it is pointed out that the direct measurement of reaeration by the tracer technique is much preferred over the indirect calculation arrived at by measuring other factors in the stream. This is particularly true since the phenomenon of biological extraction can take place, which would make the deoxygenation rate or BOD removal rates invalid. In this particular study, the rate coefficients which would be used for prediction of waste assimilation in the South River would be those determined by tracer measurement on reaeration and the deoxygenation rates subsequently calculated from those measurements. The tracer technique for measuring reaeration is a significant contribution to the sanitary engineering field and provides a tool for more accurate determination of the assimilative capacity of streams.

Reaeration Studies of the Chattahoochee River

J. R. Wallace

The question that originally concerned us on our studies of the Chattahoochee River was whether or not we could take the techniques that we had developed for small streams and apply them to a much larger stream. Through our experience with this river we can now affirmatively answer the question. Yes, we can use our methods on a larger river with some modifications. However, the modifications are not large ones. To give a little information for comparative purposes, I'll give you typical flows for the rivers we studied. Flows in the Flint River were typically 10 to 20 cubic feet per second; on the South River the flows were between 50 and 100 cfs, and on the Chattahoochee we are working with flows around 1000 to 2000 cfs.

The dose procedure is quite similar in all cases. The amount of radioactive material is, by necessity, increased as we go to the larger flows. Our license calls for a maximum dose of 5 curies of tritium and 2.5 curies of krypton on the Chattahoochee. These tracers are combined with approximately 9 liters of dye and placed in a 9 liter container. This can be compared to the 1 liter container we were using on the smaller rivers. We showed earlier that the 1 liter bottle is placed between a couple of angle irons, and we simply hit it with a hammer to make the release to the stream. We don't do this with the larger dose. We have a steel basket into which the larger bottle is placed. We place the basket with the dose into the river and, when everyone is clear of the dose point, we break the bottle with dynamite caps which we have strapped to the bottle. The dosing rig (the basket) is made so that we don't get any splash. This procedure gives us very close to an instantaneous dose. During the early part of the Chattahoochee study we were placing the dose container in the river from a boat. The dose was suspended from a steel cable stretched across the river; in addition, the dose was bouyed up by a float to which it was attached by a length of rope. This was the method we used during the first few doses on the Chattahoochee. Since this time we have decided that it is not necessary to have an individual or a couple of individuals in the boat trying to handle this rather heavy weight. For this reason we have gone to a clothesline type of rig where we put a pulley on both sides of the river and attach the dose to the line and pull it out to the desired location in the center of the river and then lower it below the surface. Then by means of a firing line attached to the rope we set off the blasting caps and the dose is released to the river.

At the sampling station the only change from the operation on the smaller rivers was in the manner of positioning the pump. On the Chattahoochee the sampling stations were at bridges, and the submersible pump was attached to a float, which in turn was held in place by

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a rope attached to the bridge. The electrical line which supplies power to the pump is connected to the rope.

As you can see, we do have a different method of release for the Chattahoochee doses; we also do some things differently as safety precautions. For example, we are never in the water with the dose as we were in the smaller streams. We always have our radiological safety officer present when we are handling this quantity of radioactive material. The series of doses that we have made has been marked by only one incident that caused us to cancel a dose. At the beginning of one dosing operation we opened the barrel in which the container was shipped, and we saw that the level of the dye had gone down in the container. At that time we were using a boiling flask with a long neck on it, and we saw the level of the dye was not completely filling the neck. We immediately took precautions to prevent any exposure. During shipping a crack had developed in this bottle, so we discharged the contents into the river, and did not try to conduct a study with this dose. As you know, a boiling flask bottle has quite thin walls. Since that time we have been using a much heavier bottle. We send the bottles along with the dye to the firm from which we buy the radioactive tracers, they put the dye in the bottles and then they insert the tracers. They then send them back to us in a mixture all ready to be put into the stream.

The other downstream procedures are similar to what we have in smaller rivers. There is, of course, more distance between sampling stations. The sampling stations were on the order of 2 miles apart on the Flint and South River. We need to use several times this distance on the Chattahoochee in order to get accurate results. Our results are not as accurate if we have only 1 percent gas loss as they are if we have a larger percentage gas loss. For this reason we are using several miles between sampling stations.

The hydraulic studies on the Chattahoochee were similar to the hydraulic studies on the Flint and the South. We are making cross section measurements every 500 feet, except this time the measurement of discharge, stream area, etc., have to be made from a boat. Here again we use standard USGS procedures for making flow measurement from boats. We are finding the hydraulic properties in the Chattahoochee River to be less variable than they were in the Flint and South, and I might add that the studies are done at a flow rate which is controlled by Georgia Power Company at a point upstream from Atlanta where they have a small hydroelectric facility. All of our Chattahoochee studies have been done on the weekend, and during the weekends the power company has maintained a constant discharge through their generators of 1000 cfs. Studies on the Chattahoochee are not complete, for various reasons; one of which being that Georgia Power Company had some work to do on their hydroelectric plant at the time we were going to finish up our series of studies last winter. The studies were delayed and the wet season caught us before we completed the series of studies. We will finish the studies on the Chattahoochee during this summer. Values for the doses that we have made on the Chattahoochee indicate

K_2's in the range of .01 to .04, values which are considerably lower than those reported on the Flint and South. These would be in terms of K_2 per hour. We will in one of the sessions tomorrow compare K_2's that we have measured with those given by some of the prediction equations and will have a little more to say about it then. Are there any questions about our study on the Chattahoochee?

Discussion

Question: Are you going to conduct studies at higher flows, or do you intend to use only low flow, around 1000 cfs, in the Chattahoochee? (JRW) We intend to work with that figure. We, at this point, are not going to try to run these tests at specific high or low rates; what we are trying to do is stay generally with low values. As was indicated earlier we are trying to get values of K_2 that will be of value not only to our research but that will be of value to some of the agencies that want these values and these are some of the flows that they are most interested in.

Question: (inaudible) (JRW) If I understood you, you asked for the average flow velocity. We are talking about 2 ft. per sec. in the Chattahoochee with average depths at a cross section of 3 to 4 ft. Locally cross sections would be deeper than that but these are average depths. So we are talking about a channel maybe 200 ft. wide, 3 ft. deep, and a velocity of 1-½ to 2 ft. per sec.

Question: What does it cost for a Chattahoochee dose? (JRW) The tracer material cost is about $850, field personnel about $250, and maybe another $200-250 for lab analysis and computation. This does not include the cost of my time nor that of Dr. Tsivoglou nor does it include any equipment costs.

Question: Do you assay your dose before releasing it? (JRW) Yes, we assay it when we get it, and take another sample before it is released. In the smaller runs we were specifying 1.0 and 0.5 curies of the respective tracers and we were typically getting around 0.7 curies, total.

Question: What are the limitations of your license to release radioactive tracers? (JRW) Our license allows us to release a specific dose in a specific river. If we want to make a dose at some other location we have to get an amendment to the license.

In obtaining the license we have to specify what uses are made of the waters down the stream; but I might add that by the time the material has gone just a few hundred yards the radiation levels are down to a point that are less than the lifetime exposure level and so it is not really a problem.

Question: How much distance do you cover with a release? (JRW) In the order of 8 or 9 miles, something like that.

Model Study of Reaeration Capacity of the James River Estuary (Virginia)

M. W. Lammering

Introduction

In order to develop a mathematical model to predict the effects of waste discharges to an estuary, information must be obtained on the mixing and dispersion characteristics of the estuary and, in the case of oxygen-demanding wastes, the reaeration capacity. Avenues of investigation open to the actual measurement of dispersion characteristics include the use of organic dye and radioactive tracers in the estuary itself and/or a hydraulic model of the estuary. Until recently, reaeration capacity, expressed as a reaeration rate coefficient(s), was computed by mass balancing techniques for oxygen and empirical correlations (1). A relatively new method involves the use of a gaseous radioactive tracer, krypton-85, for the direct measurement of oxygen transfer across the air-water interface (2).

The advantages of a hydraulic model study are economies in time, material, and personnel. Due to compressed scales (model to prototype), dispersion and reaeration measurements can be conducted over the entire estuarine system for a period corresponding to a month during an actual time of about one day. In most cases, it would not be practicable, or even possible, to carry out the same intensive study on the prototype. The disadvantages of a hydraulic model study are inherent in the design and construction of the model for uses other than the determination of mass transfer coefficients. Mass transfer similitude does not exist, thereby presenting the problem of determining the proper scaling relation between mass transfer coefficients (diffusion and reaeration rate coefficients) for the model and those for the prototype.

The following sections of this paper describe the radio-tracer reaeration study conducted in the hydraulic model of the James River estuary. This study was conducted to obtain the reaeration data required by the Middle Atlantic Region, Water Quality Office, for their mathematical model of the estuary. The hydraulic model is located at the U.S. Army Engineers Waterways Experiment Station, Vicksburg, Mississippi. Corps of Engineers personnel operated the model during the study.

Description of Hydraulic Model

The hydraulic model of the James River estuary covers a surface area of about one-half to three quarters of an acre (estimated from dimensions of the shelter housing the model). The bed of the model is

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concrete with copper "roughing" strips strategically located to yield proper salinity distribution and velocity profiles. Scaling factors (model to prototype) are as follows:

Horizontal : 1:1000
Vertical : 1:100
Velocity : 1:10
Time : 1:100
Discharge : $1:10^6$

Tidal cycles (semidiurnal tides) are sinusoidal with a time period of 7-1/2 minutes per cycle. The volume of water in the model during its operation is on the order of approximately 3×10^5 liters.

As shown in the schematic plan of the model (figure 1), the limits extend from Richmond, Virginia, to the Atlantic Ocean. In addition to the James River, fresh water inflows to the system include the Appomattox, the Nansemond, and the Chickahominy Rivers. During the reaeration study, the simulated flow in the James River was maintained at 3200 cfs. Simulated flows in the tributary streams were:

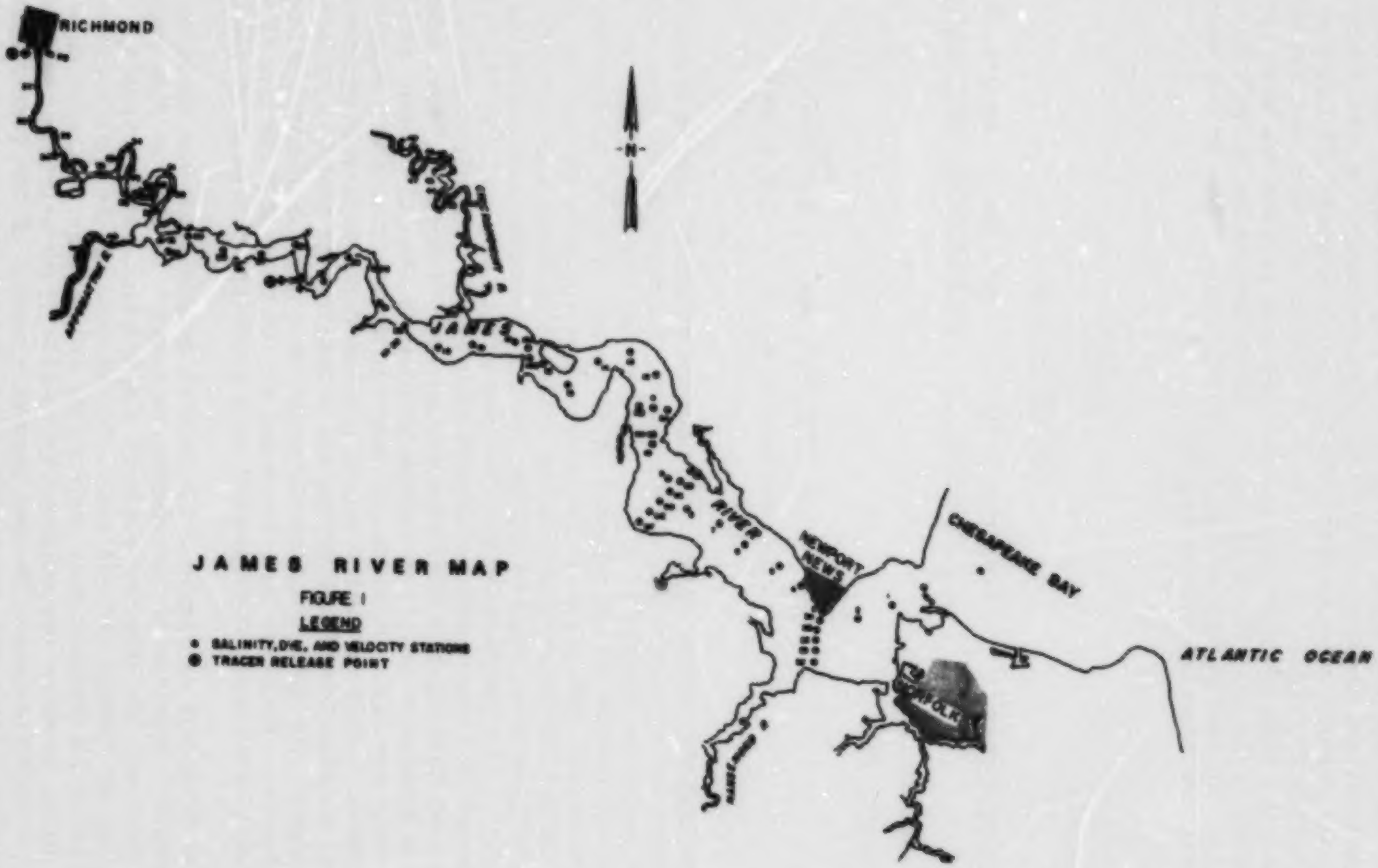
Appomattox River - 152 cfs
Nansemond River - 89 cfs
Chickahominy River - 56 cfs

Simulated waste discharges from the cities of Richmond and Hopewell were not included in the reaeration study.

Reaeration Study Procedures

To obtain reaeration data for the model reach of interest, City of Richmond to Dancing Point (Figure 1), the alternative methods for injecting tracers into the model were an instantaneous release and a continuous discharge. Both methods would have yielded the temporal distributions required for calculating diffusion and reaeration rate coefficients. The method of instantaneous release was selected on the basis of the relative ease with which it could be performed and the limitation of the model study to a maximum duration of 120 model tidal cycles (60 model days) or 15 actual hours. A continuous dosing method requires that the discharge continue until steady state conditions of constant concentrations (at a given tidal stage) are observed at each location of interest. There was doubt that such a condition would be met within the imposed time limit. To extend the study beyond 120 tidal cycles was not considered practicable in terms of the additional data that would have been obtained versus the added costs and personnel requirements.

In contrast to a true instantaneous release, dose solution was injected into the model at a uniform rate over one tidal cycle (centered about low water slack). This same procedure was used by O'Connell and Walter (3) for their dye dispersion studies in the hydraulic model



JAMES RIVER MAP

FIGURE I

LEGEND

- • SALINITY, DYE, AND VELOCITY STATIONS
- ◉ TRACER RELEASE POINT

of San Francisco Bay. To ensure complete coverage of the reach of interest and temporal distributions characterized by well-defined peaks, tracer releases were made at two locations: Richmond and Station 25 (Figure 1). The dose at Station 25 initiated the study and the tidal counter whereas the Richmond release was made during tidal cycle 18. The time lag was introduced as an attempt to minimize the overlapping or merging of the separate tracer masses at stations downstream from Hopewell.

Table I

Dose Solutions

Release Point	Fluorescein Dye (grams)	Tritium (millicuries)	Krypton-85 (millicuries)
Richmond, Va.	-	6	4
Station 25	4	-	4

Total Volume of Dose Solution - 2.2 liters

Volume Released into Model - 1.2 liters

The amounts of tracers in the two dose solutions are presented in Table I. As indicated, the conservative tracer in the dose solution released at Station 25 was fluorescein dye whereas tritium was used in the dose solution released at Richmond. The use of different conservative tracers was to facilitate data analysis at stations for which temporal distributions from the separate releases were measured. Dose solutions were prepared in the field in the afternoon preceding the study. The preparation involved the addition of tritium (as tritiated water) or fluorescein dye and krypton-85, in form of clathrate contained in a capsule (the capsule and the clathrate dissolve after a few minutes in water, releasing krypton-85 into solution), to approximately one liter of distilled water in a two-liter glass reagent bottle. After the water volume was carefully adjusted such that there was no significant air bubble after sealing the bottle with a rubber stopper, the dose solution was mixed by a magnetic stirrer for several minutes. Discharge into the model at the rate of about 1.2 liters per model tidal cycle corresponded to a prototype discharge of about 0.5 mgd.

The frequency of sample collection at each station was based on the relative location of the station in respect to the two dose release points and the expected shape of the temporal distribution(s) - a broad or sharply defined peak. In the case of the dose solution containing fluorescein dye, the arrival and passage of tracer mass past a given location was followed by on-site fluorometric measurements of dye concentrations. The net seaward movement of the tracer mass containing tritium and krypton-85 was estimated from a prior dye tracer study conducted by the Corps of Engineers at the same fresh-water inflow condition. However, this movement was also detected in a qualitative manner in the model by a counting assembly consisting of an end-window Geiger-Muller tube and scaler. When the tube was positioned just above the water surface (figure 2), the gamma-ray emissions from krypton-85 were detected. The maximum frequency of collection, every cycle (low water slack, determined by visually observing the lack of movement of floating particulates) for 10 to 20 tidal cycles following dose release, occurred at stations in the immediate vicinity of the release locations. Once the sampling of a station was initiated, sampling was generally continued until the termination of the study irrespective of the likelihood for measuring significant tracer concentrations. To cover all the stations of interest (Stations 36, 34, 33-19, 17, 15, and 53) over the study period of 120 tidal cycles, a sampling crew of eight was employed.

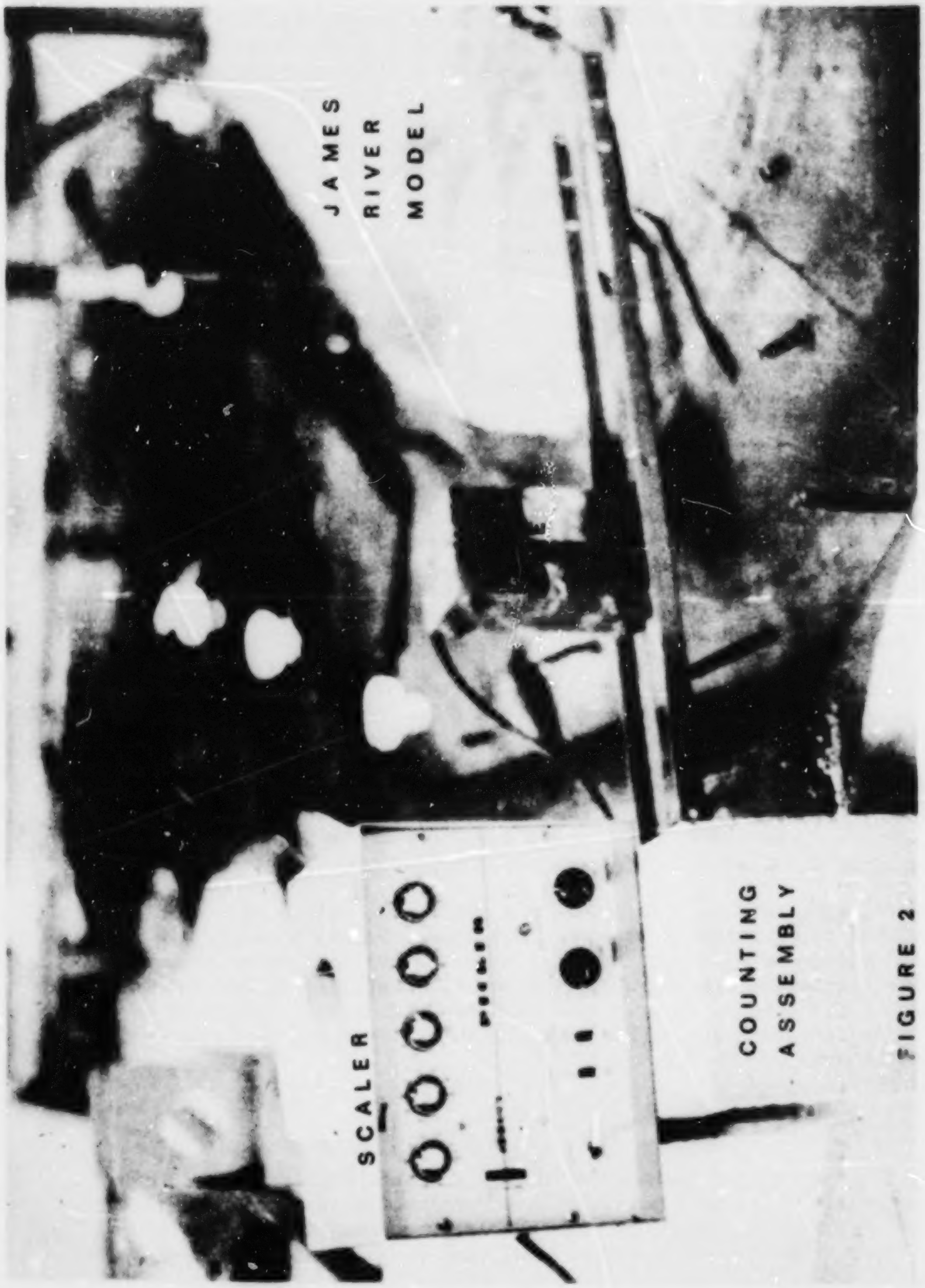
As mentioned above, water samples were collected from the model at low water slack. The collection procedure involved gently dipping a 25 ml glass vial into the water at a position corresponding to the location of the main river channel (marked on the model at each station). Considering the size of the sampling vial (7/8" dia., 3-5/16" length) in terms of model scaling factors and the prototype volume represented by each sample (about 6600 gallons) great care had to be excercised to prevent significant surface disturbance other than that caused by tidal action. The vials were fitted with "poly-seal" tops to prevent the presence of an air bubble in the collected sample.

Analyses of the dye, tritium, and krypton-85 concentrations in the water samples were performed in the laboratory of the Radiological Activities Section (Cincinnati, Ohio). The specific details of determining the tritium and krypton-85 concentrations by liquid scintillation counting have been discussed in the paper by Mr. R. J. Velten. Fluorescein dye concentrations were determined fluorometrically with a G. K. Turner Associates' Model 111 Fluorometer (lower limit of detectability of 0.5 ppb).

Water temperature in the model was monitored continuously at Station 28 and intermittently at other locations. Unlike salinity, velocity, and tidal elevation, no controls were exerted over water temperature. The simulated James River inflow was substantially cooler than the bulk of the model water. Also, as the outdoor and indoor air temperatures increased during the day, the water temperatures in the model increased.

Method of Data Analysis

Following an instantaneous or nearly instantaneous release into a hydraulic model or estuary, a conservative tracer (tritium or dye) is dispersed by turbulent mixing whereas gaseous krypton-85, a nonconservative tracer, undergoes the same physical dispersion as well



COUNTING ASSEMBLY

FIGURE 2

as loss to the atmosphere by diffusion across the air-water interface. The back and forth motion of the tracer mass caused by tidal action yields temporal distributions (concentrations as a function of tidal cycle) at each station which show build-up to maximum values followed by die-off as the tracer mass passes through the area. In this respect, the advantage of releasing the dose solution over a tidal cycle is that the temporal distributions can be integrated to yield steady state concentrations (3). Despite the fact that the estimation of steady state concentrations was not an objective of the reaeration study, it was considered desirable to obtain the necessary data in the event a future need should arise.

The ratio of the krypton-85 concentration to the conservative tracer concentration as a function of time exhibits a decreasing trend at a given sampling station. This is due to the loss of krypton-85 to the atmosphere as the tracer mass passes through the area. Thus, the rate constant for krypton-85 loss can be calculated from the temporal distributions for the station by the following relationship:

$$\ln (C_{Kr}/C_{CT})_t = (A)e^{-(K_{Kr})(t)}$$

where A = constant

C_{Kr} = krypton-85 concentration at time, t

C_{CT} = tritium or dye concentration at time, t

t = time

The constant, A, does not have to be evaluated since a plot of the logarithm of (C_{Kr}/C_{CT}) versus time (plotted as tidal cycle-low water slack) yields a straight line of slope, K_{Kr}. Calculated by this technique, the rate constant represents an average value for a volume element of the model or estuary. Although it remains to be determined by mathematical analysis, the length of the volume segment is, at least in part, a function of the magnitude of the tidal excursion in the immediate vicinity of the sampling station. Since the sampling stations in the James River model are separated by distances less than tidal excursions, the volume segments represented by temporal distributions for adjacent stations were overlapping in the reaeration study.

As reported by Tsivoglou (2, 4), the reaeration rate constant for oxygen, k_2, is calculated from the corresponding K_{Kr} by dividing by the factor of 0.83. Research has shown that this correction factor is apparently applicable in all situations.

Results

Examples of typical temporal distributions for krypton-85 and tritium or dye at several locations in the model are shown in Figures 3, 5, 7, 9, and 11. At each location, except Station 53 in the Curles Neck, the distributions exhibited a buildup in concentration to a maximum value and a die-away edge skewed to the right. (The radioactivity data plotted in terms of activity units, cpm, actually refer to activity per 2 ml of sample volume. For fluorescein dye, 1 ppb was arbitrarily set equal to 1 cpm per 2 ml.) This pattern was not observed at Station 53 because sampling was not started in time to measure the leading edges of the tracer distributions. The krypton-85 distribution for Station 23 (Figure 9) is of particular interest since it shows the cumulative effect of the mixing of the leading edge of the dispersed tracer mass released at Richmond with the trailing edge of that released at Station 25. As shown, both releases contributed to the instantaneous krypton-85 concentrations after tidal cycle 35.

An interesting observation was the time lag between the peak concentrations of krypton-85 and tritium at locations substantially downstream from the tracer release point (Richmond). For example, at Station 29 the peak krypton-85 concentration was observed at tidal cycle 31 in comparison to tidal cycle 37 for tritium. At locations close to the tracer release point, Stations 32 and 34, the peak concentrations were coincident in time. The time lag was observed because the temporal distributions were for a fixed point in space. If spatial distributions had been determined, such a time lag between peak concentrations would not have occurred. Although beyond the scope of this paper, it can be shown that the dispersion coefficient can be calculated from the magnitude of the measured time lag.

For all main channel locations (except Stations 30, 31, and 32) and Station 53 in the Curles Neck, the semi-logarithmic plot of concentration ratio (krypton-85 to conservative tracer) versus tidal cycle was well fitted by a straight line (Figures 4, 8, 10, and 12). In the case of locations such as Station 23 where the data from both doses were used to obtain independent measurements of the rate constant for krypton-85 loss, the plotted values of C_{Kr}/C_{H-3} were calculated from krypton-85 data corrected for the tracer release at Station 25. This involved a point-by-point subtraction of the extrapolated portion of the distribution attributable to the Station 25 dose (Figure 9). The plots of concentration ratio versus time for Stations 30, 31, and 32 could not be fit with a single straight line, but required two lines with substantially different slopes. As indicated in Figure 6, the line of greater slope was designated "Mode 1" and the other "Mode 2". Although this author is unable to provide a positive explanation for this finding, it is assumed that "Mode 2" reflected the slow bleeding of tracers from the Curles Neck and the Appomattox River into the main channel after the bulk of the tracer mass had passed downstream.

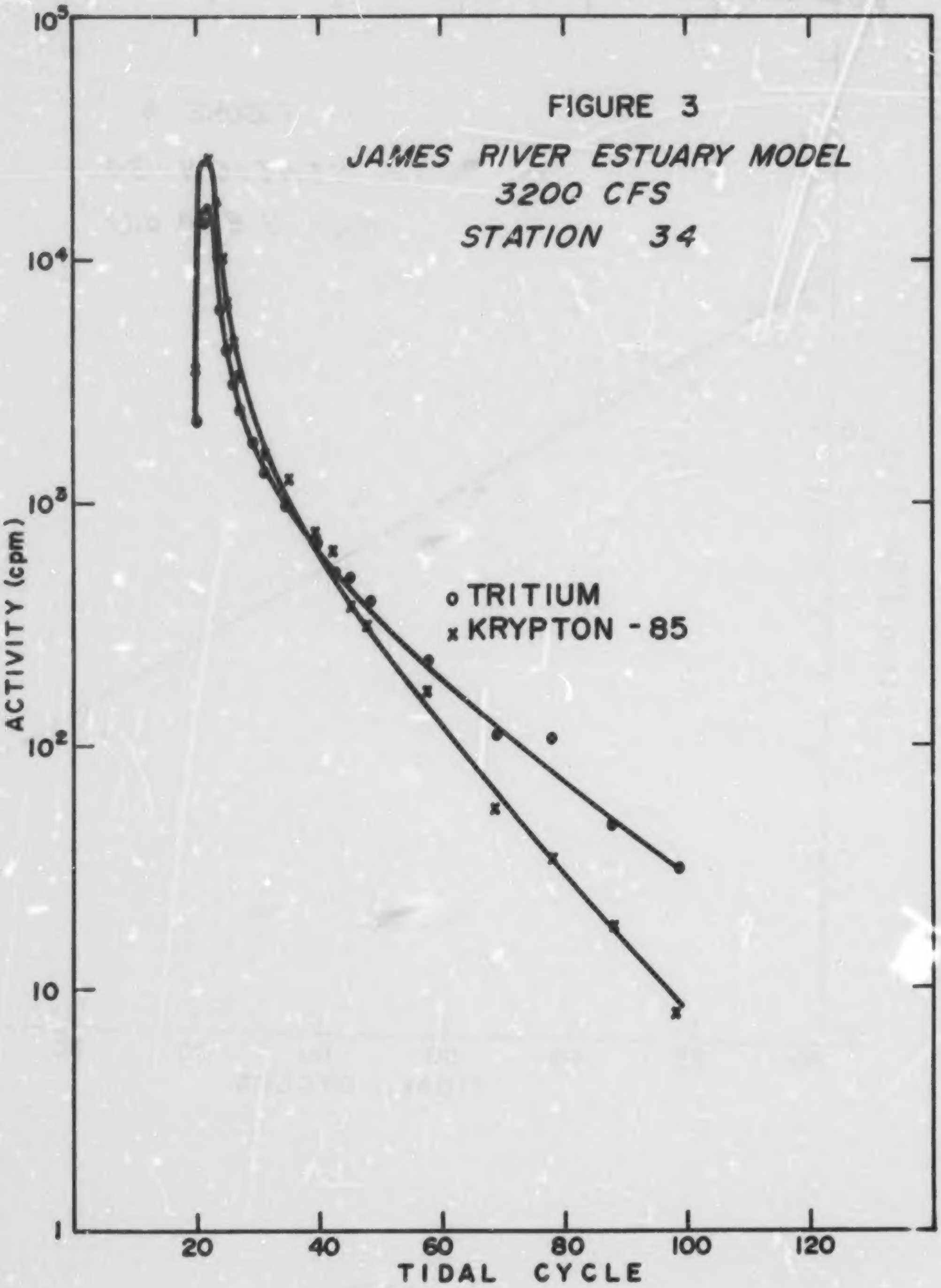


FIGURE 3

JAMES RIVER ESTUARY MODEL
3200 CFS
STATION 34

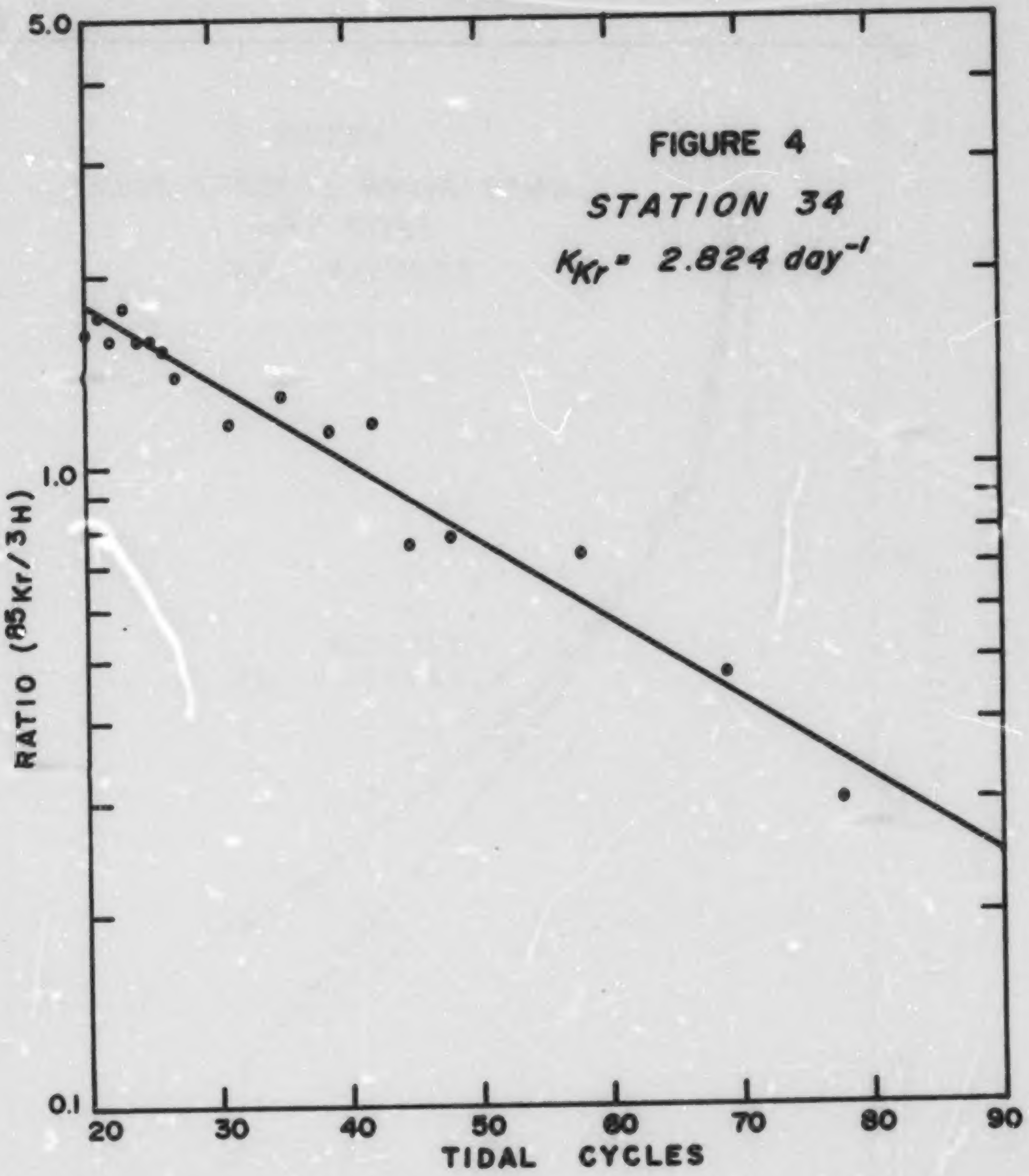


FIGURE 4
STATION 34
K_{Kr} = 2.824 day^{-1}
5.0
1.0
0.1
RATIO (^{85}Kr/^{3}H)
20
30
40
50
60
70
80
90
TIDAL CYCLES

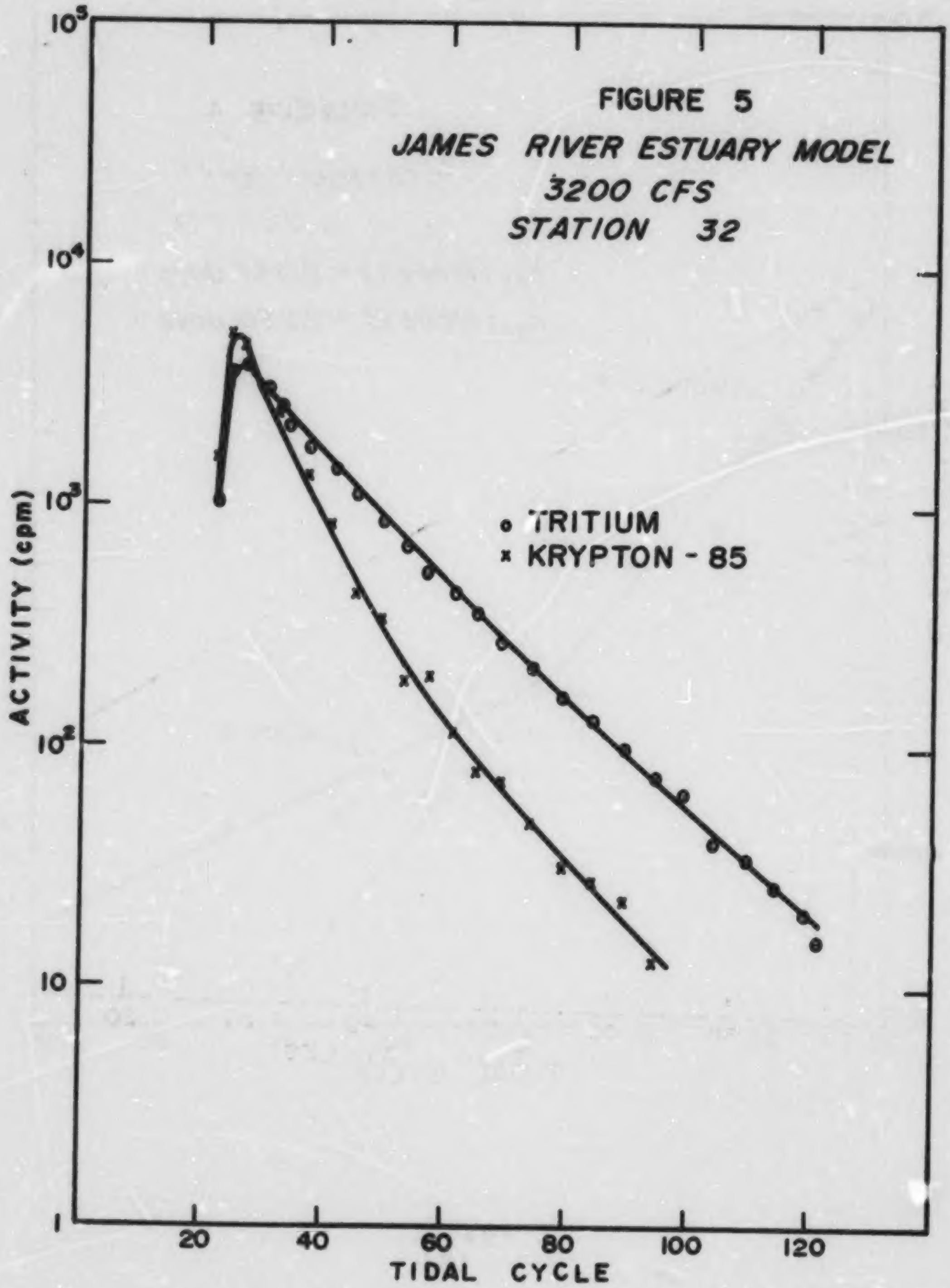


FIGURE 5
JAMES RIVER ESTUARY MODEL
3200 CFS
STATION 32

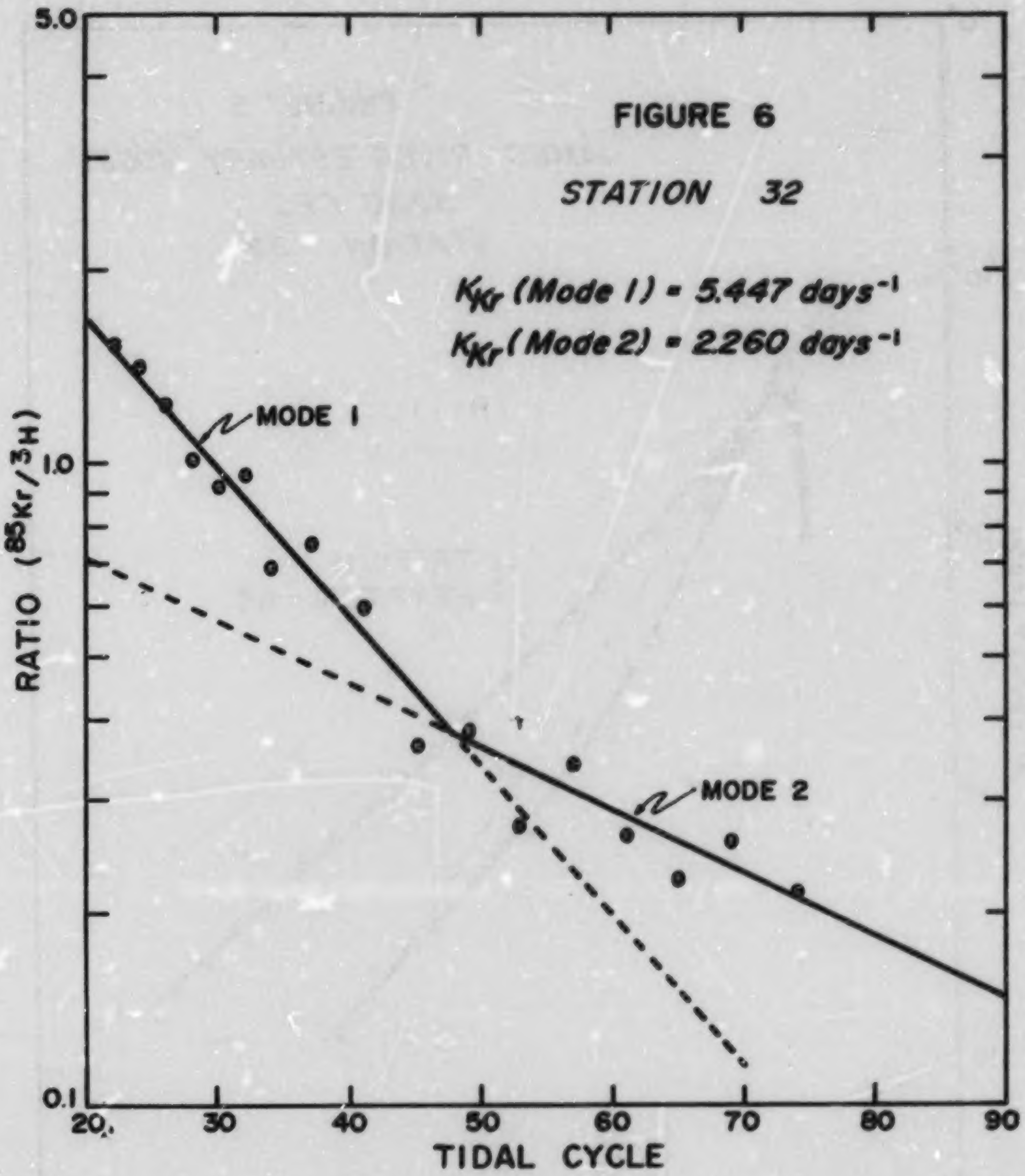


FIGURE 6

STATION 32

K_{Kr} (Mode 1) = 5.447 days^{-1}

K_{Kr} (Mode 2) = 2.260 days^{-1}

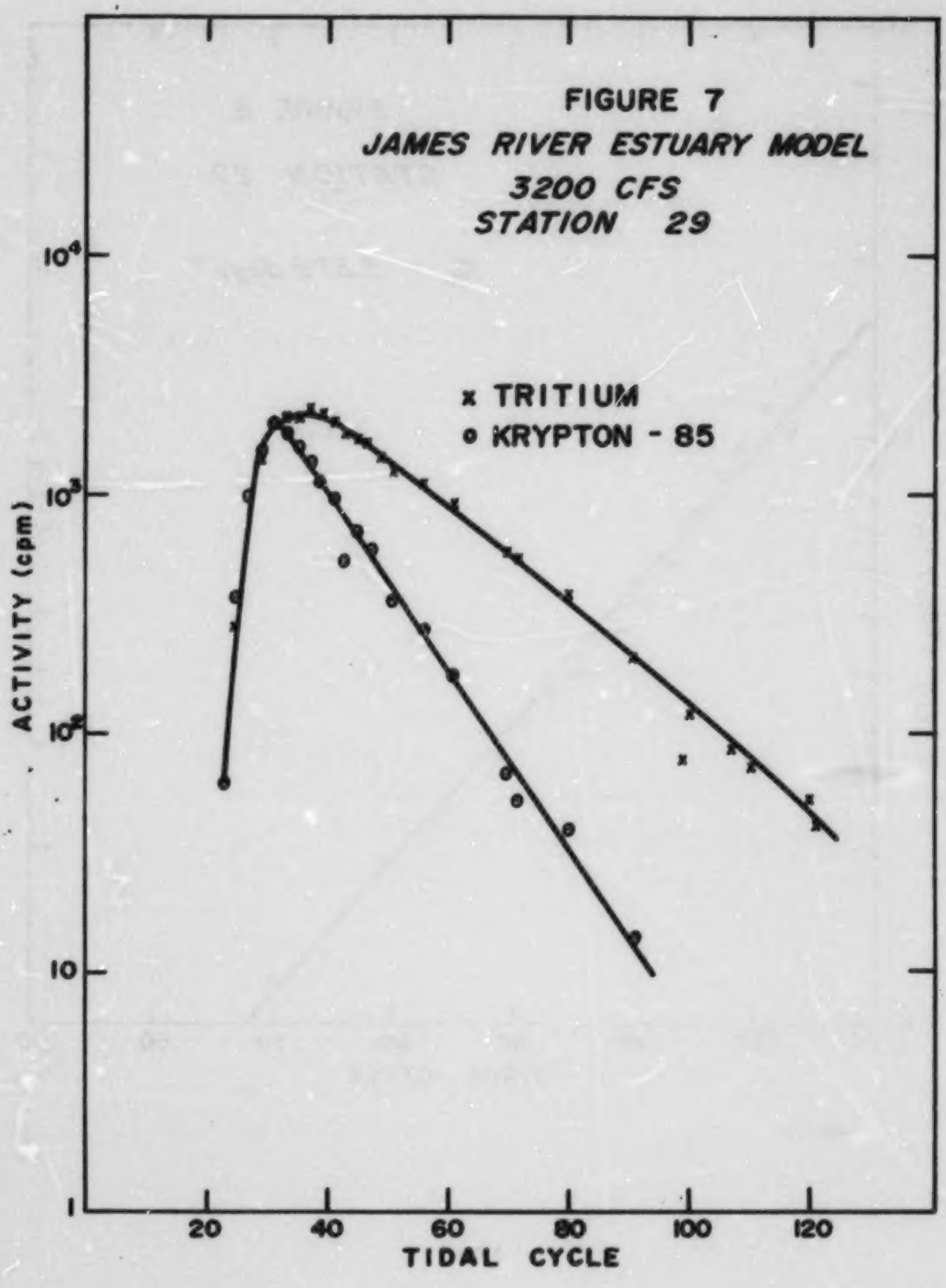


FIGURE 7

JAMES RIVER ESTUARY MODEL

3200 CFS

STATION 29

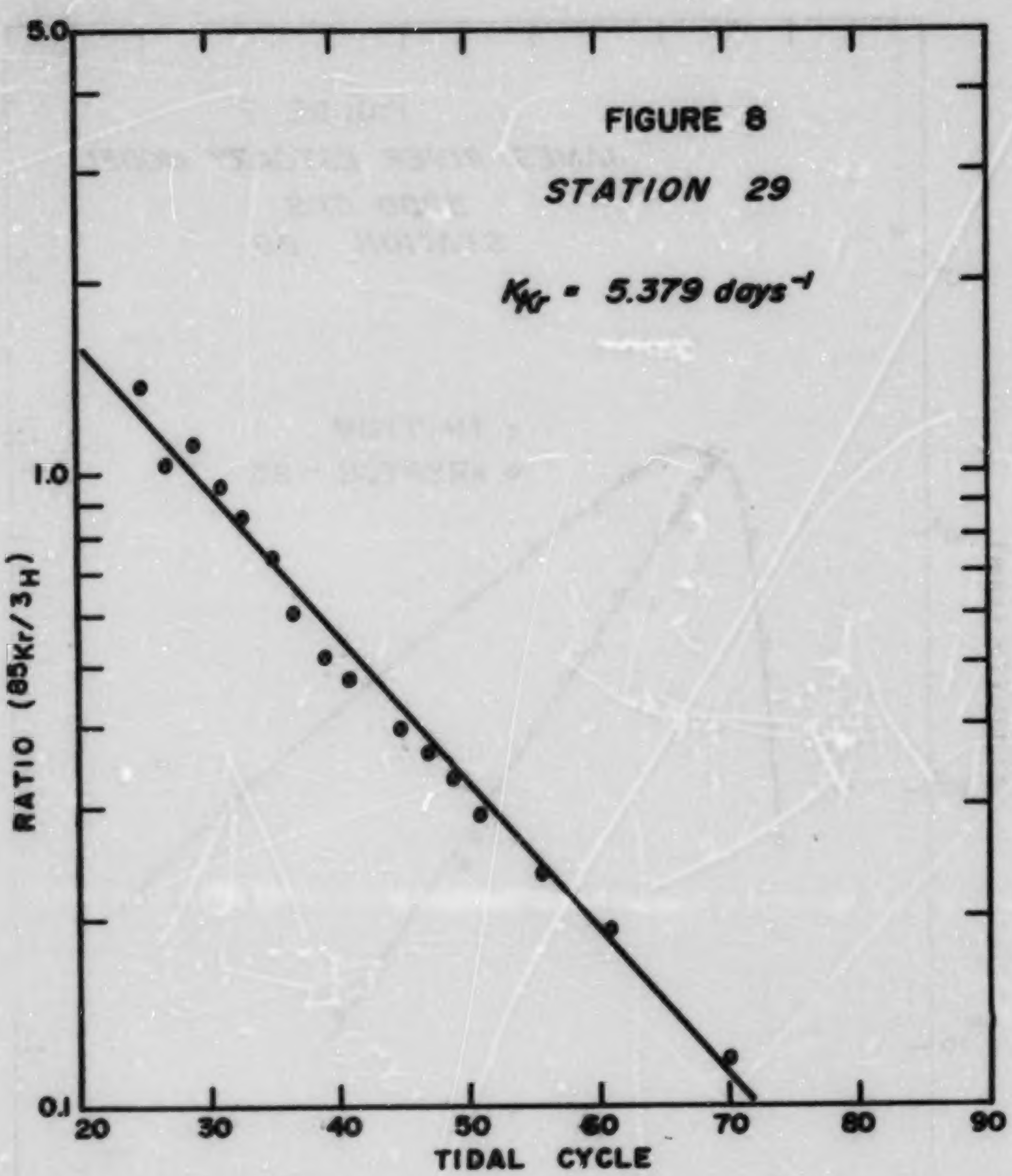


FIGURE 8
STATION 29
K_{Kr} = 5.379 days^{-1}
5.0
1.0
0.1
RATIO (^{85}Kr/^{3}H)
20
30
40
50
60
70
80
90
TIDAL CYCLE

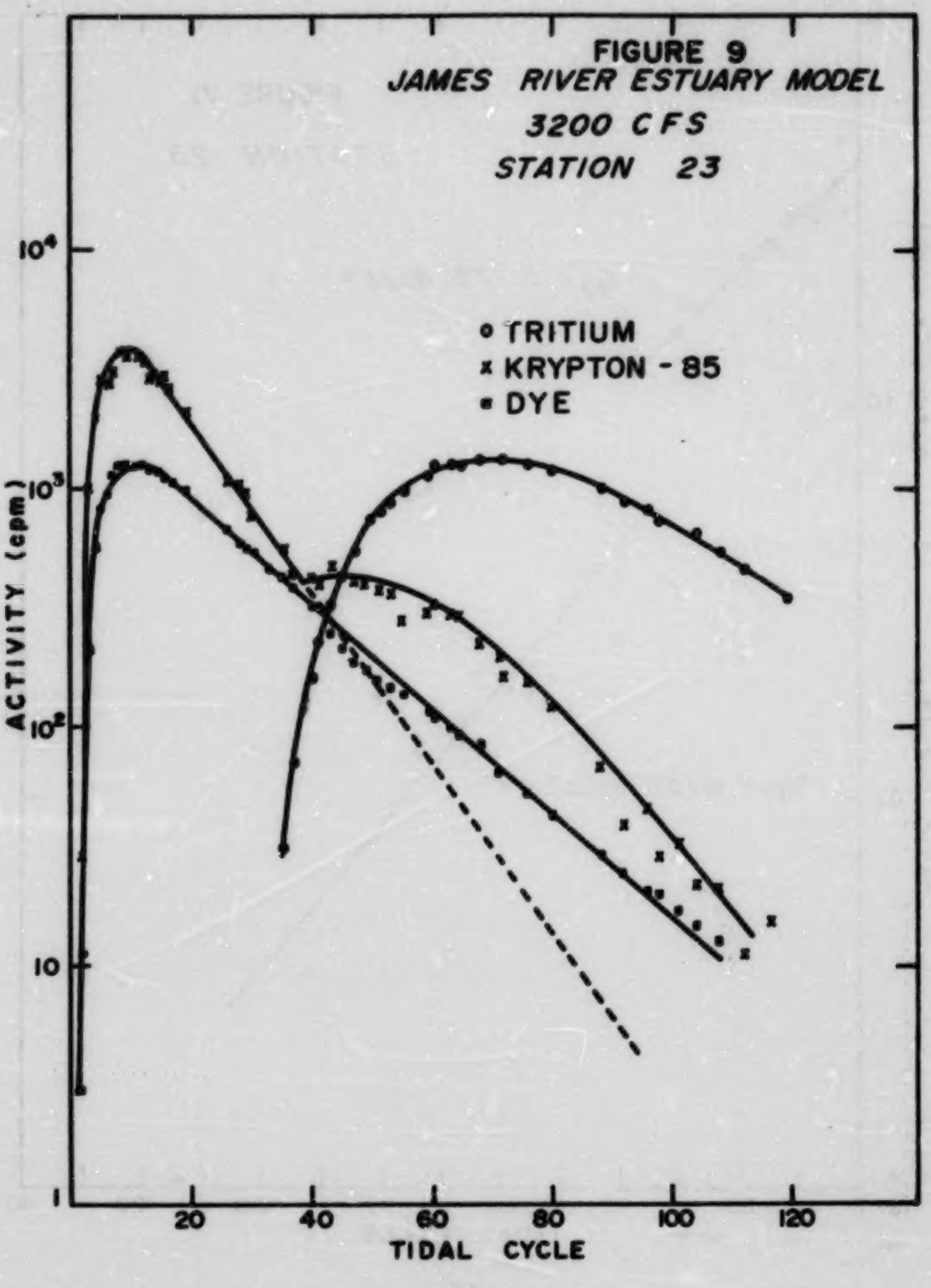


FIGURE 9
JAMES RIVER ESTUARY MODEL
3200 CFS
STATION 23

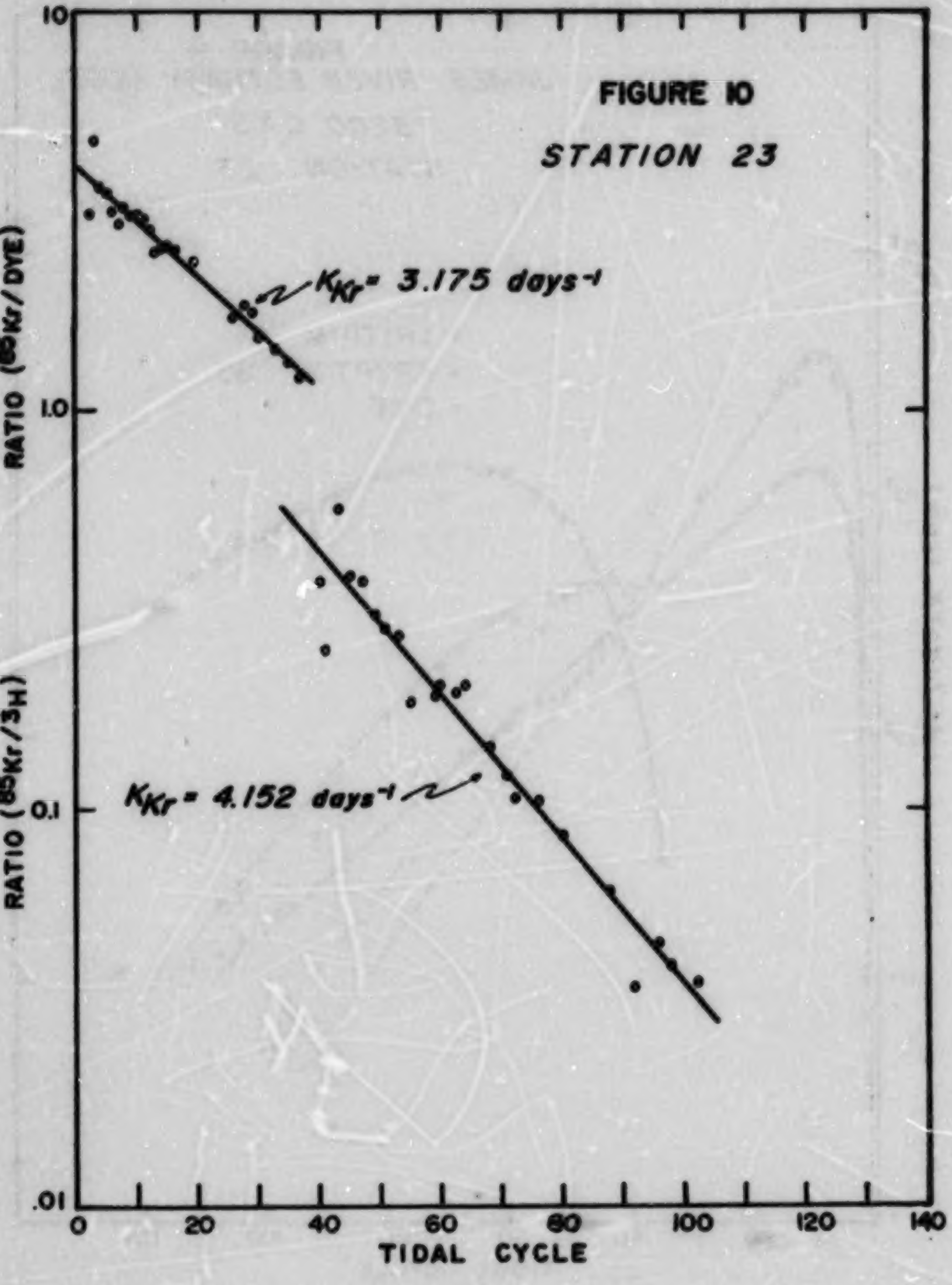


FIGURE 10

STATION 23

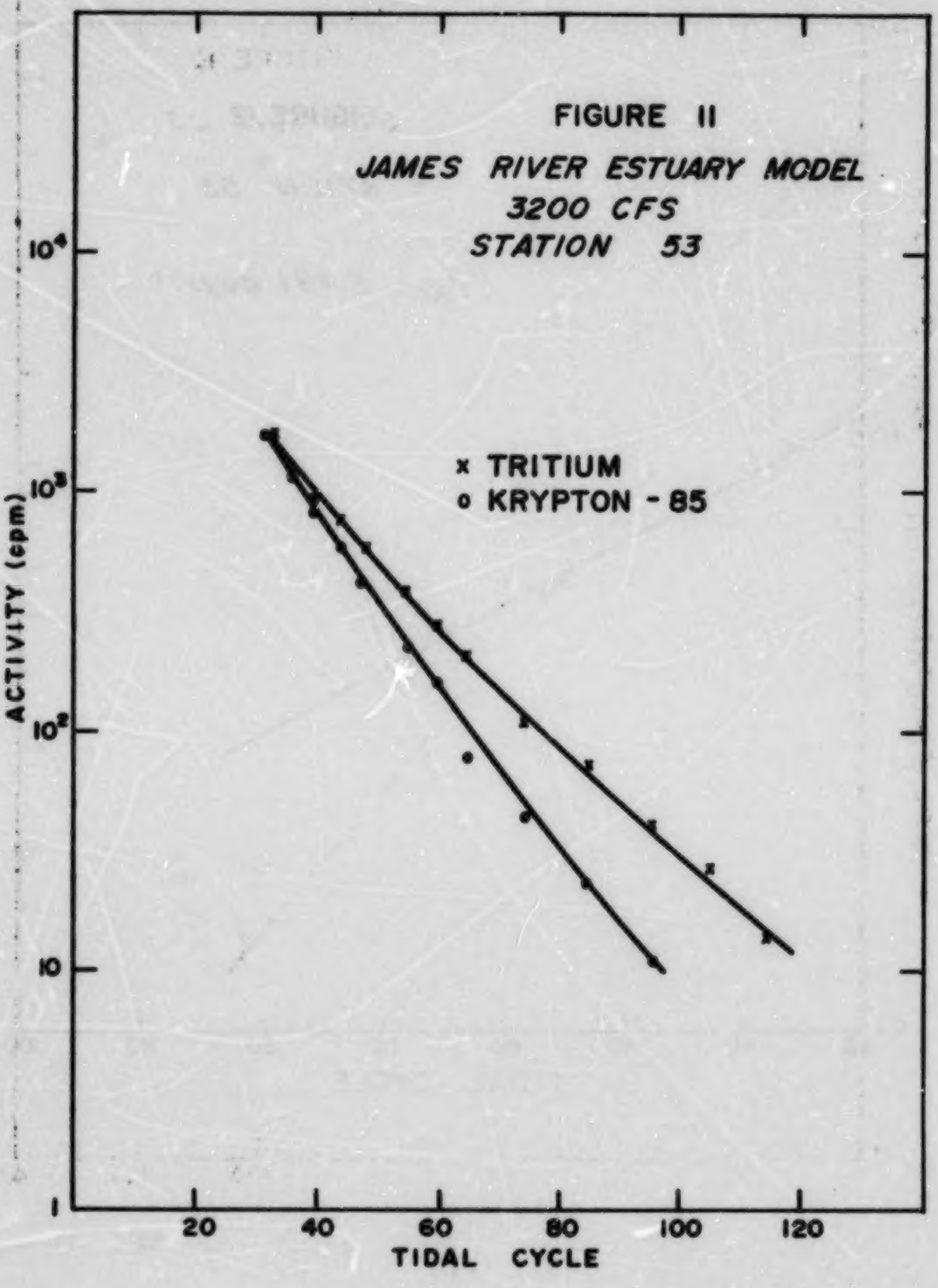


FIGURE II

JAMES RIVER ESTUARY MODEL
3200 CFS
STATION 53

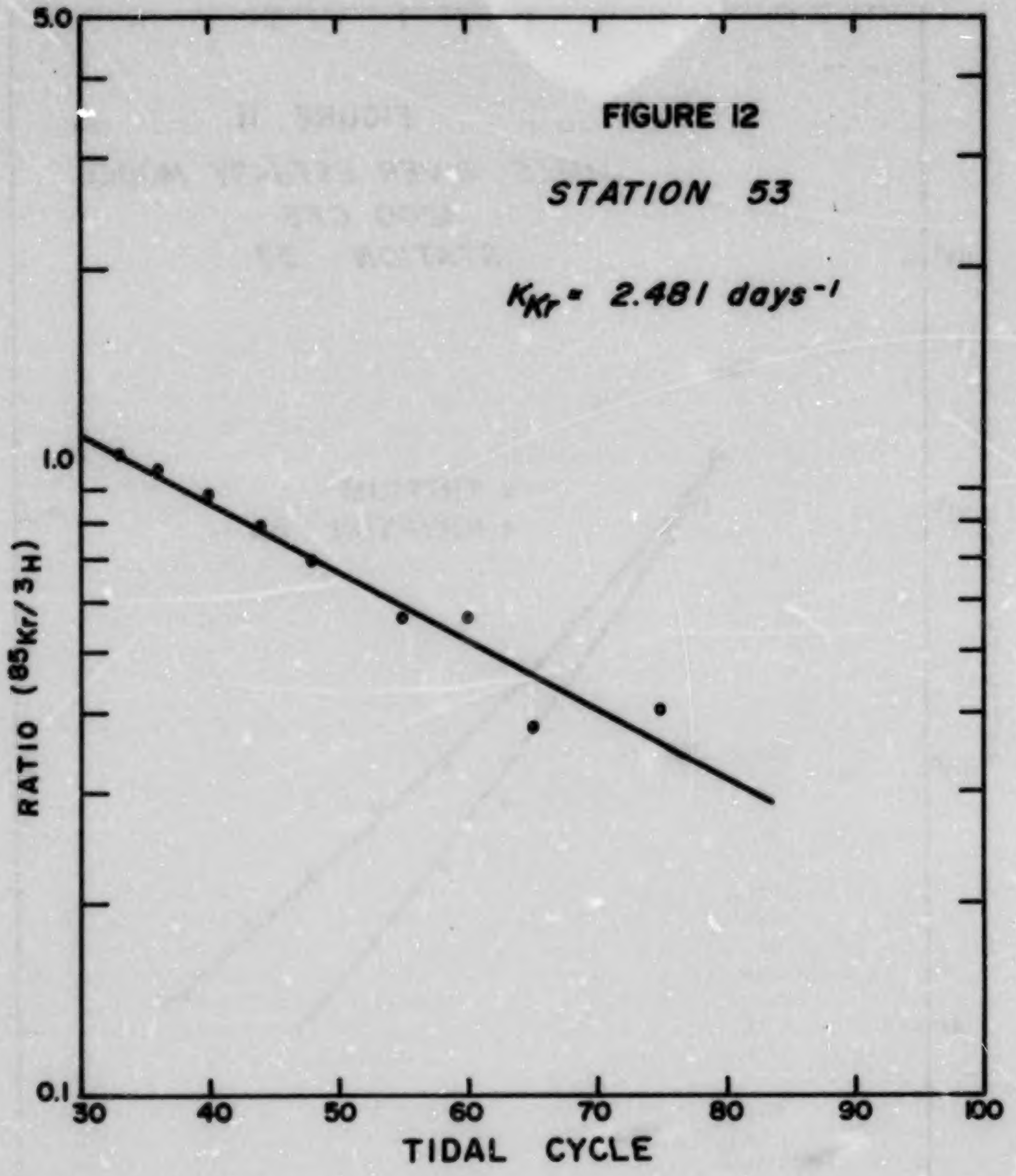


FIGURE 12
STATION 53
K_{Kr} = 2.481 days^{-1}
5.0
1.0
0.1
30
40
50
60
70
80
90
100
TIDAL CYCLE
RATIO (^{85}Kr/^{3}H)

The calculated rate constants for krypton-85 loss, K_{Kr}, and reaeration, k_2, are summarized in Table II. The close agreement between the independent measurements at Stations 27 through 23 was indicative of the relative accuracy of the study procedures, the high degree of reproducibility of the multi-tracer technique under controlled conditions and the stability of the model. Support for the assumption that "Mode 2" loss of krypton-85 at Stations 31 and 32 was representative of the sloshing action between the Curles and the main channel was indicated by the similarity in magnitude of the "Mode 2" rate constants with the rate constant for Station 53 (1.1 and 2.3 model days^{-1} versus 2.5 model days^{-1}).

Reaeration rate constants for the prototype were calculated by applying a reduction factor of 1/100th to the model values. This scaling factor was used because it represented the model to prototype ratio for time. That this was a valid approach was indicated by the apparent success with which the reaeration rate constants were used in the mathematicaly model of the prototype for oxygen-demanding wastes. Admittedly, however, the dissolved oxygen profiles predicted by the mathematical model were not influenced as greatly by variations in the reaeration rate constant as by small variations in the dispersion coefficient. An additional piece of data which supports the scaling factor of 1/100th is O'Connor's (5) estimate of the reaeration rate constant for the lower estuary in the vicinity of Hopewell. For a James River flow of 3150 cfs, O'Connor used an empirical equation to calculate a rate constant of 0.071 days^{-1} (at 25°C). This is in almost exact agreement with the average value of 0.069 days^{-1} calculated from the results for Stations 29 and 30.

Conclusions

The method involving the use of gaseous krypton-85 for the direct measurement of reaeration capacity was successfully employed in the hydraulic model of the James River estuary. Two separate releases of tracer, at Richmond, and at Station 25 in the lower estuary, provided independent measurements of the reaeration rate constant at several stations in the lower estuary. The agreement obtained indicated that the method is accurate as well as reproducible.

Evidence of a somewhat circumstantial nature indicated that the appropriate scaling factor for the conversion of model rate constants to appropriate prototype values is the scale factor for time. In the case of the James River model, this scaling factor was 1/100th. However, the question of the appropriate scaling factor remains to be an item of interest and one certainly worthy of additional research. If it can be shown that the reaeration data from a distorted hydraulic model can be accurately scaled up to the prototype, a most significant tool will have been provided to the engineer faced with the problem of predicting the impact of waste discharges into an estuary.

Table II

Rate Constants for Krypton-85 Loss and Reaeration

	Model Rate Constants (at 25°C)			
	K_{Kr} (model days^{-1})		k_2 (model days^{-1})	
Station	Mode 1	Mode 2	Mode 1	Mode 2
36	8.0		9.6	
34	2.8		3.4	
33	3.3		4.0	
53 (Curles Neck)	2.5		3.0	
32	5.4	2.3	6.5	2.8
31	5.7	1.1	6.9	1.3
30	6.0	3.9	7.2	4.7
29	5.4		6.5	
28	5.3		6.4	
27	4.9 (2.9)		5.9 (3.5)	
26	4.3 (3.7)		5.2 (4.5)	
25	2.6 (2.9)		3.1 (3.5)	
24	3.9 (3.4)		4.7 (4.1)	
23	1.6 (1.4)		1.9 (1.7)	

Values in parantheses were computed from temporal distributions produced by the tracer release at Station 25.

Acknowledgment

The generous cooperation and assistance of the Corps of Engineers personnel responsible for the operation of the hydraulic model during the tracer study is gratefully acknowledged.

Dr. E. C. Tsivoglou, Georgia Institute of Technology, provided invaluable assistance in the planning and conduct of the field study.

The personnel who worked long and hard hours in the collection of samples included Miss Audrey Donahue, Mr. J. P. Longtin, Mr. B. L. Smith, Mr. W. W. Finley, Mr. K. Ballentine, Mr. L. VanDenBerg, and Mr. R. Mierenfeld. The quality of their work is reflected in the comprehensive nature and accuracy of the results obtained.

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Field Studies in Yaquina River Estuary Of Surface Gas Transfer Rates

by

D. J. Baumgartner, M. H. Feldman, L. C. Bentsen, and T. L. Cooper

Introduction

The purposes of this study are (1) to measure a range of values for K_2 (defined in equation 1 below) in a natural estuary, using krypton-85; (2) to determine if variations in K_2 can be related to energy distribution associated with wind stress, velocity gradients in the channel and density gradients in the channel; and (3) to improve procedures for measurement of K_2 in bays and estuaries. The purpose of this paper is to provide preliminary information on accomplishments relating to field measurements.

The net surface transfer rate mechanism assumed for loss of dissolved krypton-85 to the atmosphere is

$$\frac{d[Kr]}{dt} = -K_2[Kr], \tag{1}$$

where

[Kr] = instantaneous concentration of dissolved krypton-85 in water;

K_2 = krypton reaction rate constant, variously entitled reaction velocity, and reaction rate coefficient.

Measurement of K_2 for krypton is pertinent to the subject of atmospheric oxygen transport into unsaturated liquid systems, using the ratio developed by Tsivoglou (1967), viz:

$$(K_2)_{OXYGEN} = 1.2(K_2)_{KRYPTON} \tag{2}$$

$(K_2)_{OXYGEN}$ has historically been suspected of being related to turbulence in the liquid phase, either by that name, or something akin to it, as "vertical mixing." Streeter and Phelps (1925) mentioned the relationship to "frictional resistance to flow," implying a dependence on energy dissipation. Osborne Reynolds, apparently, was the first to describe a form for the turbulence-energy dissipation function (Hinze, 1959). O'Connor and Dobbins (1956) established a relationship for $(K_2)_{OXYGEN}$

related directly to the rate of energy dissipation per unit mass of fluid, viz:

$$(K_2)_{OX} \propto \overline{U}Sg, \qquad (3)$$

where

$\overline{U}$ = mean stream velocity,
S = slope of energy gradient,
g = gravitational acceleration.

Gameson (1958) mentioned that reaeration at weirs was directly related to the distance of "free fall." A review by Thackston and Speece (1966) summarizes additional work on aeration at weirs. Rather accurate methods are available to compute the energy loss at such channel transitions.

Dobbins (1964) pointed out that vertical distributions of energy supply and dissipation are greater near the stream bottom and less at the surface, and, including other considerations of the surface transfer of oxygen, developed a relationship expressing the rate determining coefficient as a function of the energy transfer at the surface and the interfacial area. The surface area was recognized as sensitive to wind, but probably related to a Froude number, and for "most natural streams" not increased more than eight percent over the projected surface area. He did not intend to include wind energy contributions. Downing and Truesdale (1955) used a small tank to show that wind speeds increasing from 3 to 14 meters/sec (5 cm above surface) increased the mass transfer coefficient, f, where f = $[K_2V/A]$ (V = Volume of liquid participating in exchange, A = Effective surface area) by a factor of 40. Waves, superimposed independently, caused increased transfer at low wind speeds, but hindered transfer at high speeds. Waves, separately, increased transfer by a factor of nearly 20, depending on wave height and frequency. Measurements in the Thames estuary indicated an increase in f from 0.5 cm/hr to approximately 30 cm/hr for a range of surface effects due, presumably, to both wind and wave conditions. Kanwisher (1963) conducted similar tank studies and found that $[K_2V/A]$ varied over a range of nearly 30 in approximate proportion to the square of the wind speed.

Guinasso, et al. (1968) used data by Broecker (1964) on the radon-222 distribution near the sea surface to calculate a transfer coefficient 5 times greater than Kanwisher's value for a comparable wind speed. They also measured xenon-133 distributions resulting from an explosive release of two cureis of Xe-133 below the surface. A transfer coefficient calculated from xenon data was about 7 times greater than expected using Kanwisher's lab data for comparable wind speeds.

Many of these authors, and others, have recognized the importance of nonuniform, vertical density distribution in restricting general application of previous values of $(K_2)_{OX}$ for estuaries, and in use of formulae for computing $(K_2)_{OX}$ from stream depth and mean velocity; however, no new quantitative approaches have been offered for this situation. In addition, we consider surface wind and wave effects to be of sufficient importance in bays and estuaries to cast considerable doubt on the use of these formulae for K_2 calculations.

Tsivoglou's method appeared to offer a method directly applicable to field studies to develop new relationships for K_2 as a function of energy input at the surface via wind, and "hydraulic" energy terms, including vertical velocity and density gradients.

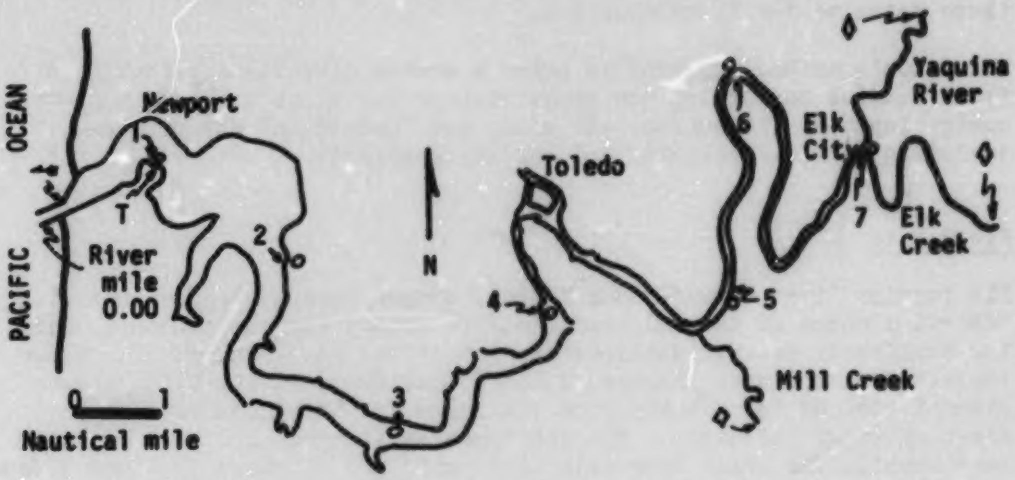
Field Site

The Yaquina River estuary near Toledo, Oregon, was chosen because it offered a range of natural conditions necessary for collection of data for completely general application, yet it was small enough to fit our logistic constraints. Figures 1 and 2 (Callaway, et al, 1970) show a general view of the estuary from the ocean to the approximate upper limit of tidal influence. The gas transfer studies conducted to date have been in the reach from mile 14 to mile 16. Figures 3, 4 and 5 show more detail of this reach. Station numbers are approximately one-half mile intervals starting with mile 14. There are no major changes in the flow direction or channel width between Station 1-4. The depth (at mean tide) decreased from 12 to 9 feet near Station 2.

Figure 6 shows a typical set of tidal conditions during a test. The stage range is about 9 feet and the current ranges from nearly +2 to -2 knots. On April 7, the current at Station 2 would be expected to have been somewhat reduced and delayed in time compared to the predicted value. The test was planned for maximum negative current and minimum $\partial V/\partial t$ at Burpee. At other times of the year the stage and current curves are phased differently so the effects of velocity and depth may be separable for the same channel configurations. We anticipate being able to conduct experiments over a range of wind values. Freshwater flows in the river range from 100 to 3000 cfs (Callaway, et al., 1970), generally high in winter and low in summer, producing a modifying effect on current and stage. The dominant effect, however, is seen in the density gradient, as high runoff tends to produce vertical stratification generally in the estuary. Tests under a variety of density distribution conditions may provide data useful for relating gas transfer rates to stream hydrodynamics. Table 1 is an example of the density distribution data collected for each run.

Procedure

Presently, current meters are set in place up-current from the injection point, a vertical profile of salinity and temperature is made, surface current drogues are released, and approximately 400 millicuries (mc) each of krypton-85 and tritium water mixed with rhodamine WT and local



Legend

○ Conductivity Meter Location
–< Wind Recorder Location
T Tide Gauge Location
◇ Stream Gauge Location

	Station	River Mile* (Nautical)
(1)	OSU Dock	~ 1.5
(2)	Sawyer's Dock	~ 3.5
(3)	Fowler's Dock	~ 7.0
(4)	Criteser's Dock	~ 9.5
(5)	Burpee	~14.0
(6)	Charlie's Dock (Fritz)	~16.0
(7)	Elk City	~19.5

* River Mile 0.00 is the seaward end of the south jetty.

FIGURE 1. FWPCA Stations, Estuary Diffusion Project, Yaquina Estuary (From Callaway et al., 1970)

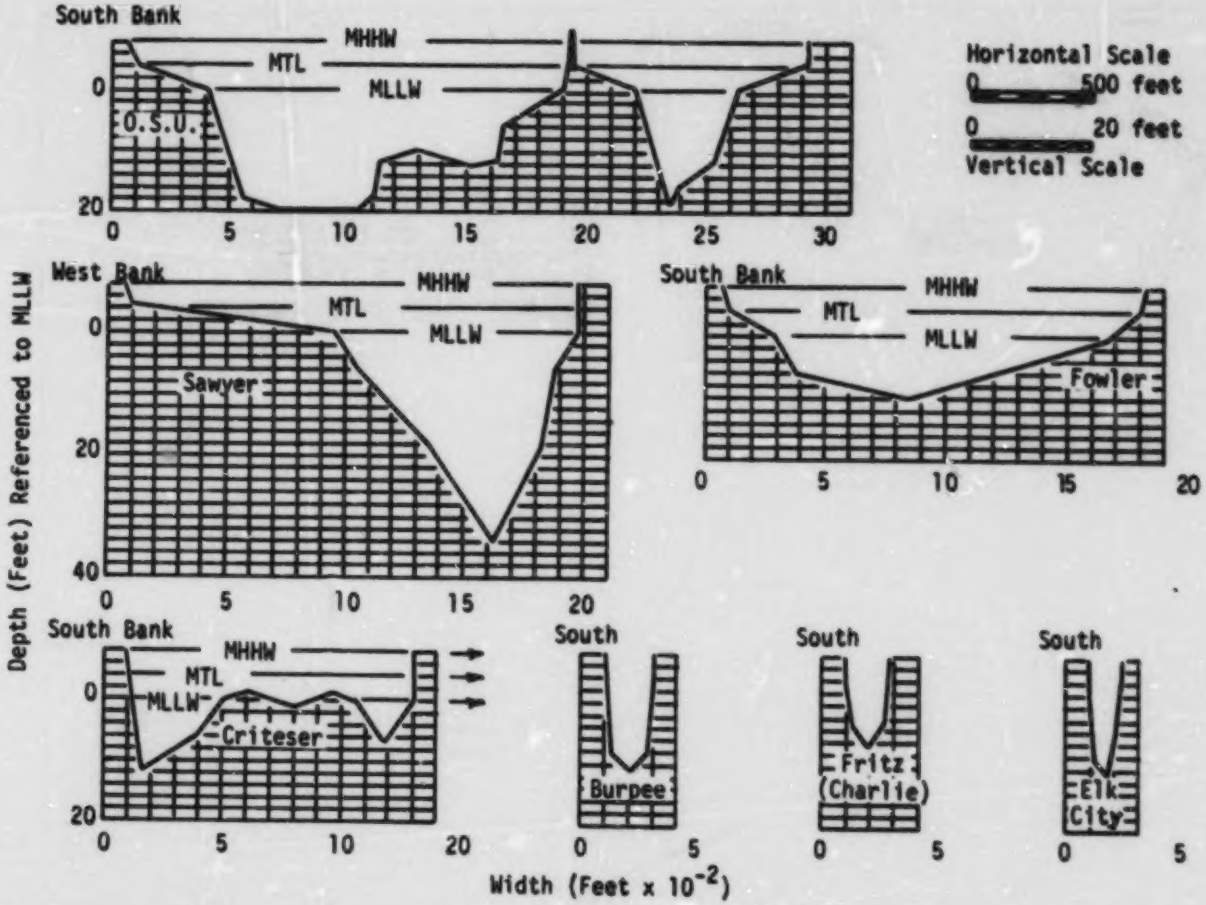


FIGURE 2. Cross Sections of Estuary at Conductivity Monitoring Sites, Yaquina Bay, Oregon
(From Callaway et al., 1970)

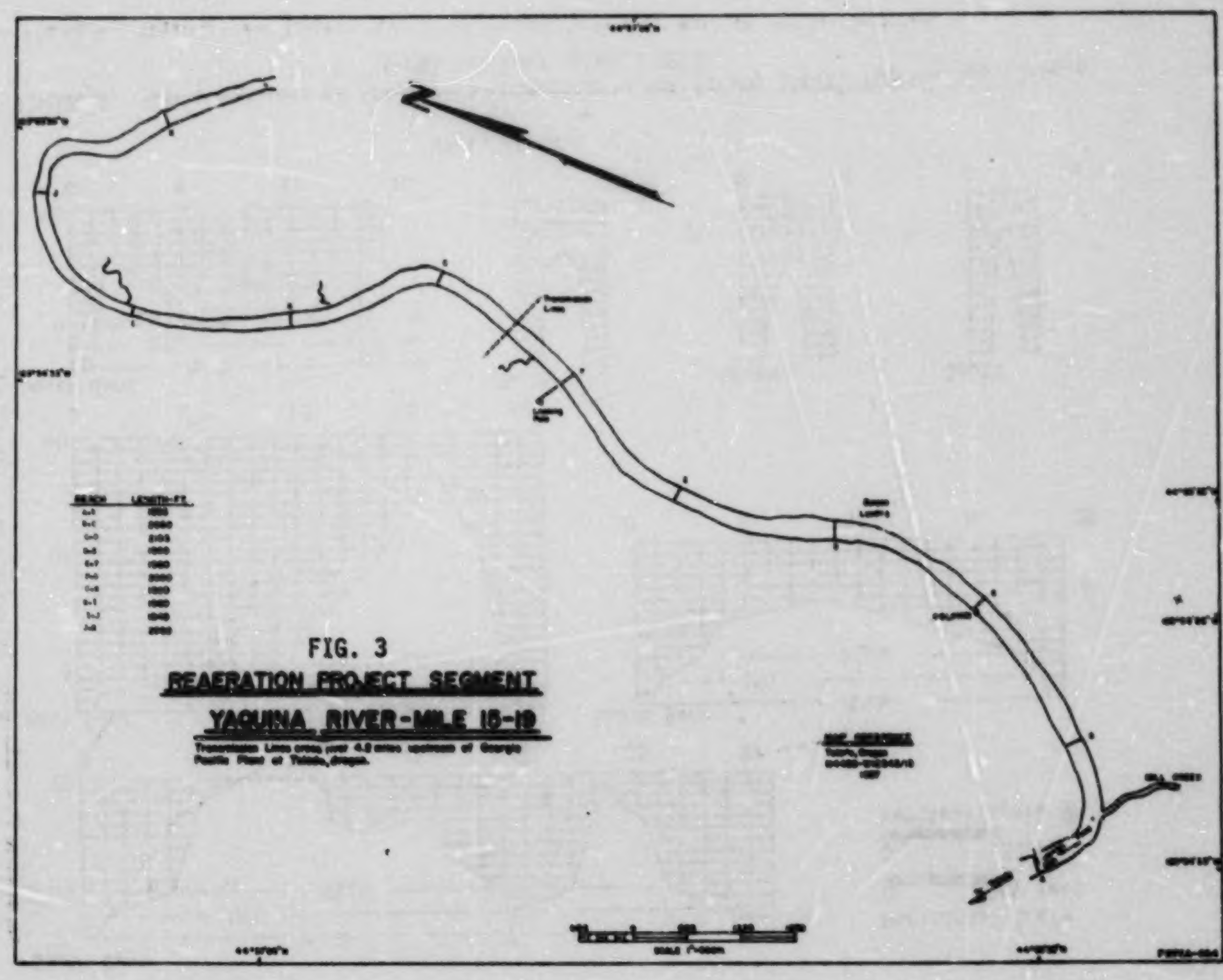


FIG. 3

REAERATION PROJECT SEGMENT

YAQUINA RIVER-MILE 15-19

Transmission Line cross over 4.0 miles upstream of Georgia Pacific Plant at Toledo, Oregon.

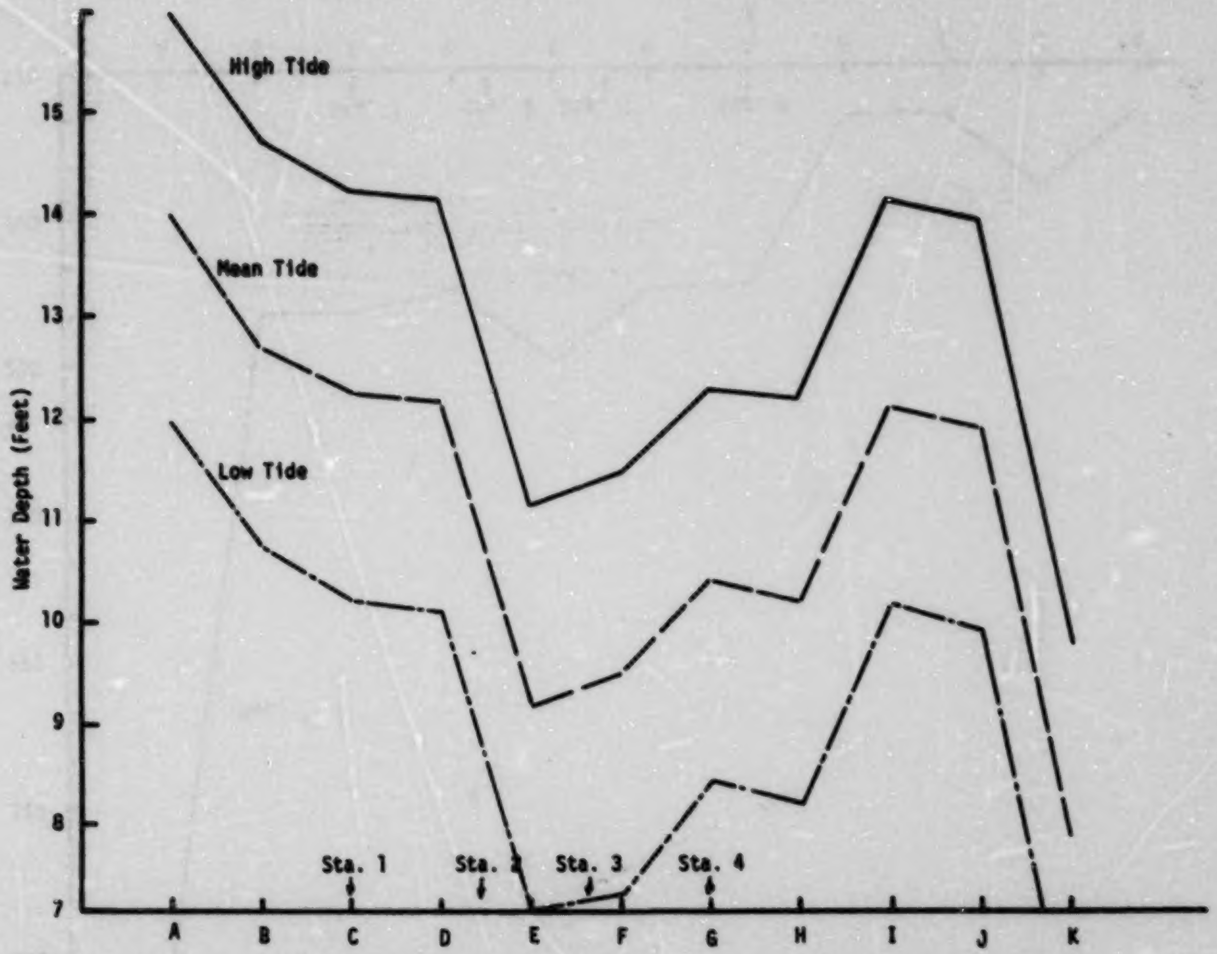


FIGURE 4. Yaquina River Estuary Reaeration Project Segment. Depth at Various Sections.

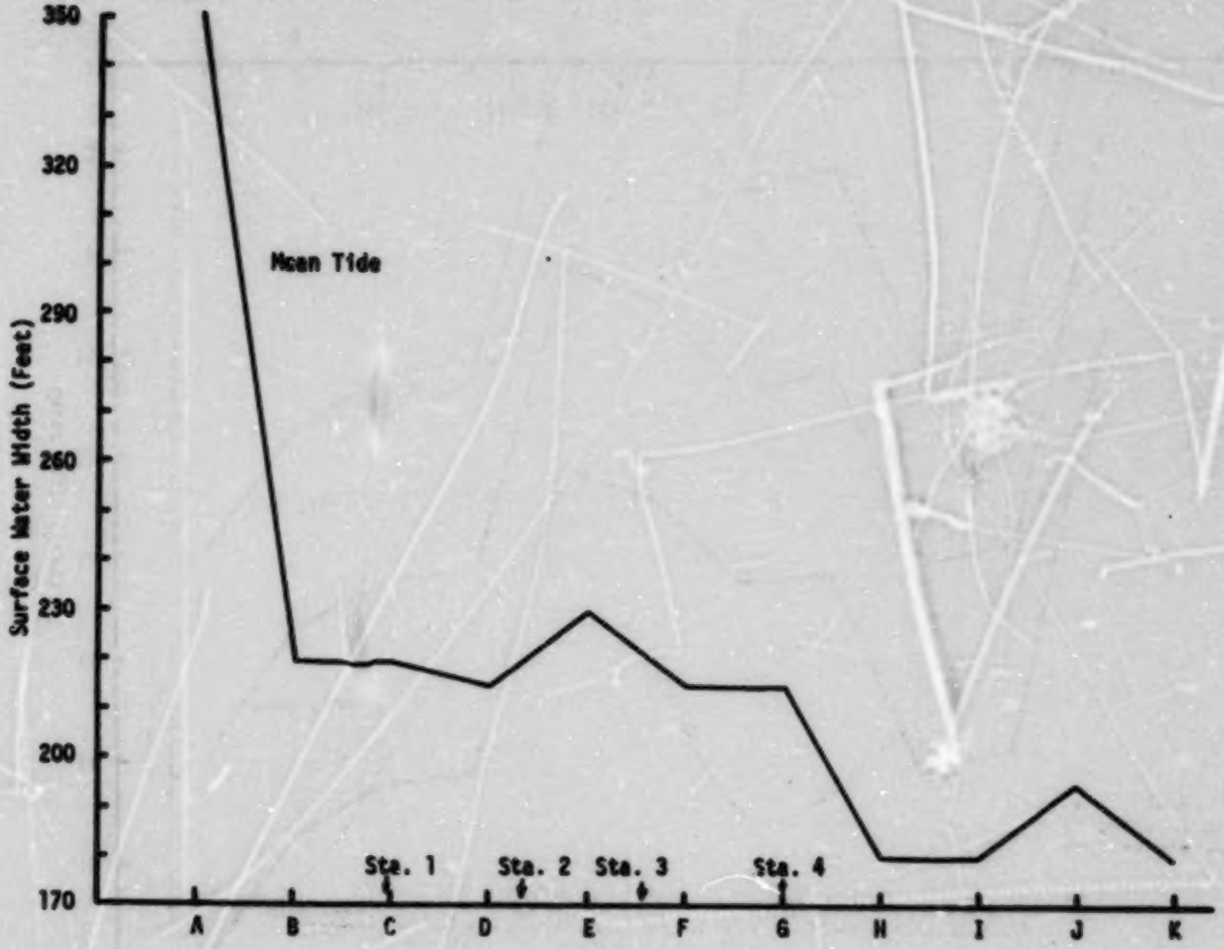
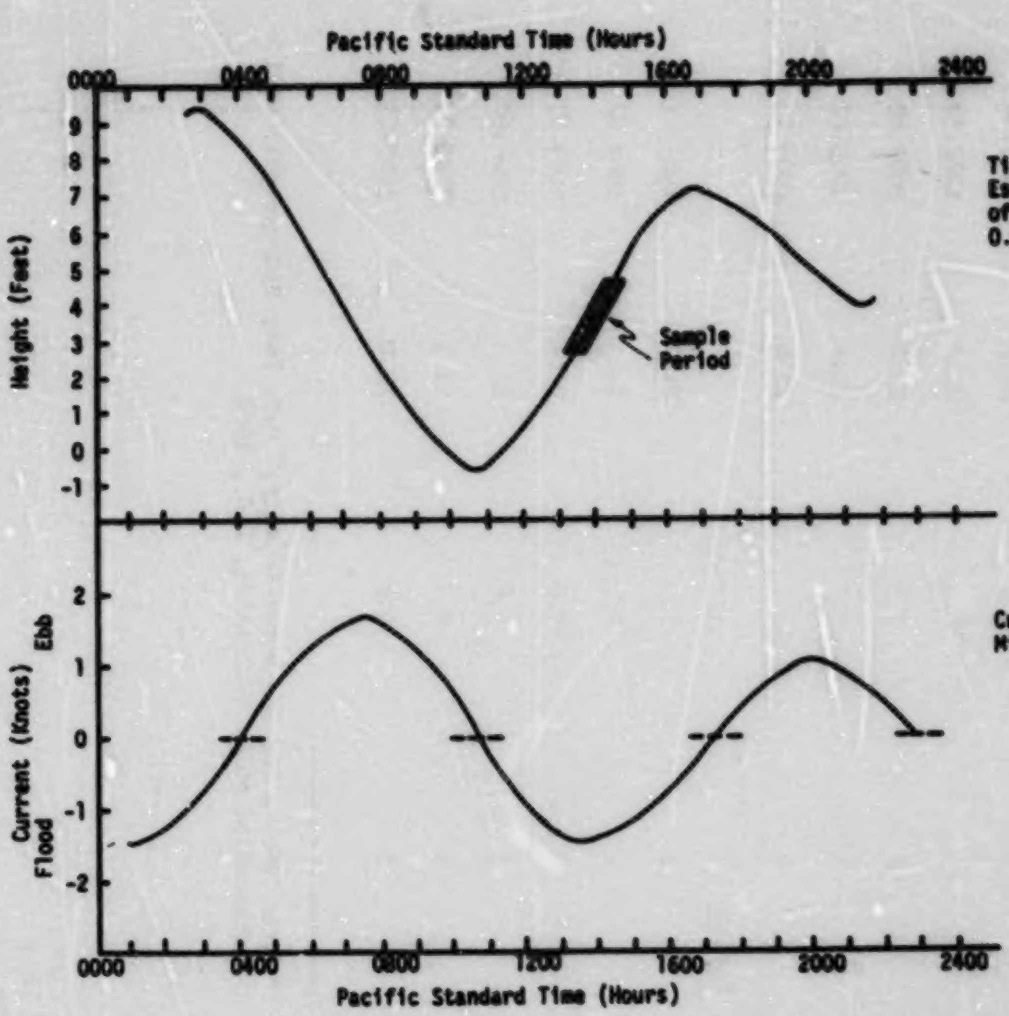


FIGURE 5. Yaquina River Estuary Reaeration Project Segment. Width at Various Sections.



Tide at Station 2 (Mile 14)
Estimated from observations
of C. Goodwin, C.E. Dept.,
O.S.U.

Current predicted at
Mile 10

FIGURE 6. Yaquina River Estuary Reaeration Project Segment. Tidal Conditions April 7, 1969.

TABLE 1

DENSITY DISTRIBUTION NEAR STATION 2, SEPT. 2, 1969

	Depth, m	S, ‰	T, °C	ρ*, kg/m^3
Start				
	0	6.1	21.2	1002.64
	1	6.3	21.2	1002.79
	2	7.1	21.2	1003.40
	3	7.7	21.4	1003.80
	4	10.1	22.0	1005.46
Finish				
	0	7.5	21.3	1003.67
	1	7.7	21.3	1003.82
	2	7.8	21.4	1003.87
	3	8.8	21.5	1004.60
	4	9.7	21.5	1005.28
	5	9.8	22.0	1005.23

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* From "Tables for Sea Water Density," U.S. Navy Hydrographic Office Publication #615, Washington, D.C., 1952.

water are discharged into the center of the stream. Distribution of tracers across the stream was used on some runs before the present procedure was adopted. Water is pumped from just below the surface (6"-12") and the discharge end of the injection hose is oriented horizontally the same distance below the surface. It is felt this will minimize density differences between the tracer solution and the ambient water, and prevent the tracer from spreading too thinly on the surface or sinking to unknown depths. The boat is driven down-current slowly past the dye to the desired sampling location. The boat is then directed up-current and held in position against the current by alternately engaging and disengaging the clutch at low throttle. A continuous stream is pumped aboard from 6"-12" below the surface, through a fluorometer, and a small stream is split from the discharge line to provide samples for subsequent analyses. The sample stream is inserted in a pre-numbered 25 ml liquid scintillation vial and slowly withdrawn when dye is detected in the fluorometer, care being taken to assure 3 volumes or more have been flushed through the vial and no gas bubbles are trapped as the polycone screw cap is securely attached. Samples are taken until the dye peak has passed, then the boat is moved to the next station. In several early trials, a second pass was made through the dye field with the sampler two feet below the surface. Generally, no dye was detected and the practice has been discontinued to minimize artificial disturbance of the tracer field. Samples are returned to the lab, transferred to replicate counting vials containing scintillation fluid and each counted twice in a Packard 3375 (Mention of proprietary names in this report is for identification only. No endorsement is implied.). Individual vial counts are tabulated, corrected for background, and tritium counts corrected for krypton "spill-over".

Natural logs of the corrected (krypton count/tritium count) ratio are computed and tabulated by a digital computer.

Results

Figures 7, 8, 9, and 10 show the results of five trials conducted since June 1969. Not all ratios obtained were plotted since many counts were judged not sufficiently above background.

Results for runs on June 17 and September 2 are plotted on the same graph (Fig. 7) to show what may have been the result of a significantly greater flooding current on June 17. Data for the run on September 9, conducted near the end of a high tide as slack current was being approached, are shown on Figure 8. Currents were greater than on September 2, and the density was vertically uniform initially, but a definite pycnocline developed between 1 and 3 meters before the run was completed. On September 17, injection was shortly after low tide as current was beginning to flood. Data are shown on Figure 9. The run on May 26 was under tidal conditions similar to those of September 17, but the wind was higher (wind speed at 22 ft.~15 k with frequent gusts to 35 k). On all runs there may have been other similarities or differences we did not measure.

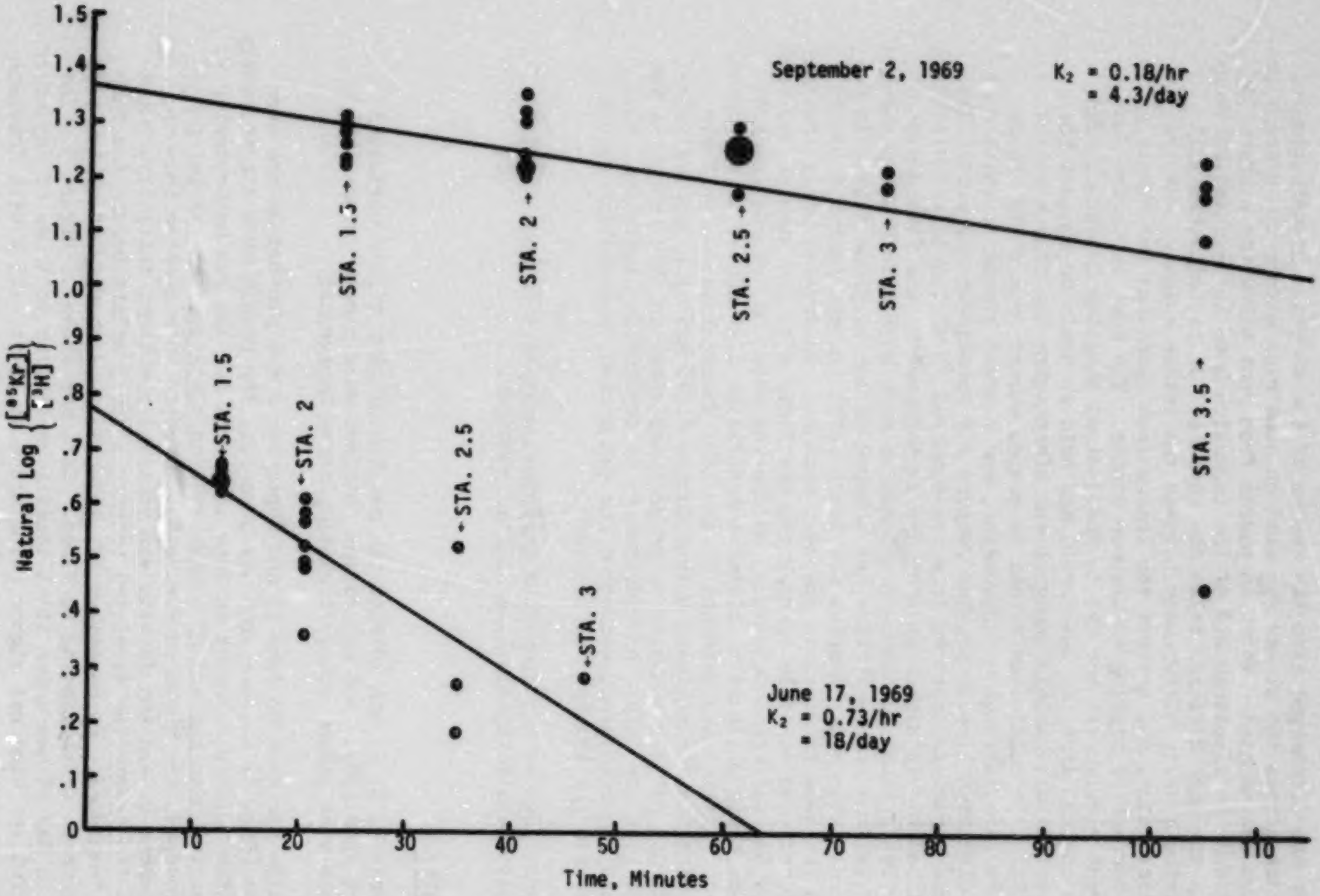
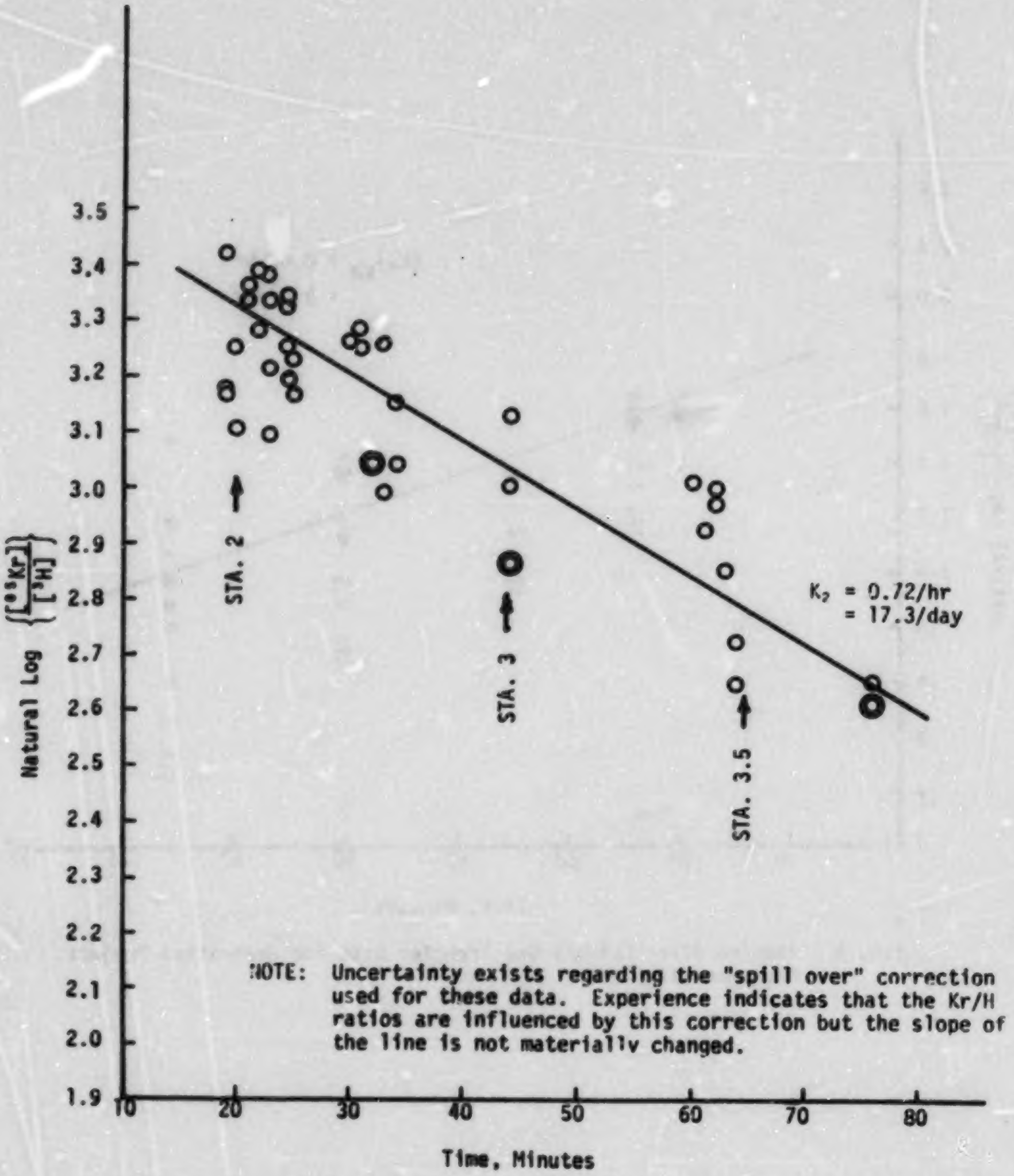


FIG. 7. Yaquina River Estuary Gas Transfer Data for Reaeration Project



NOTE: Uncertainty exists regarding the "spill over" correction used for these data. Experience indicates that the Kr/H ratios are influenced by this correction but the slope of the line is not materially changed.

FIG 8. Yaquina River Estuary Gas Transfer Data for Reaeration Project September 9, 1969

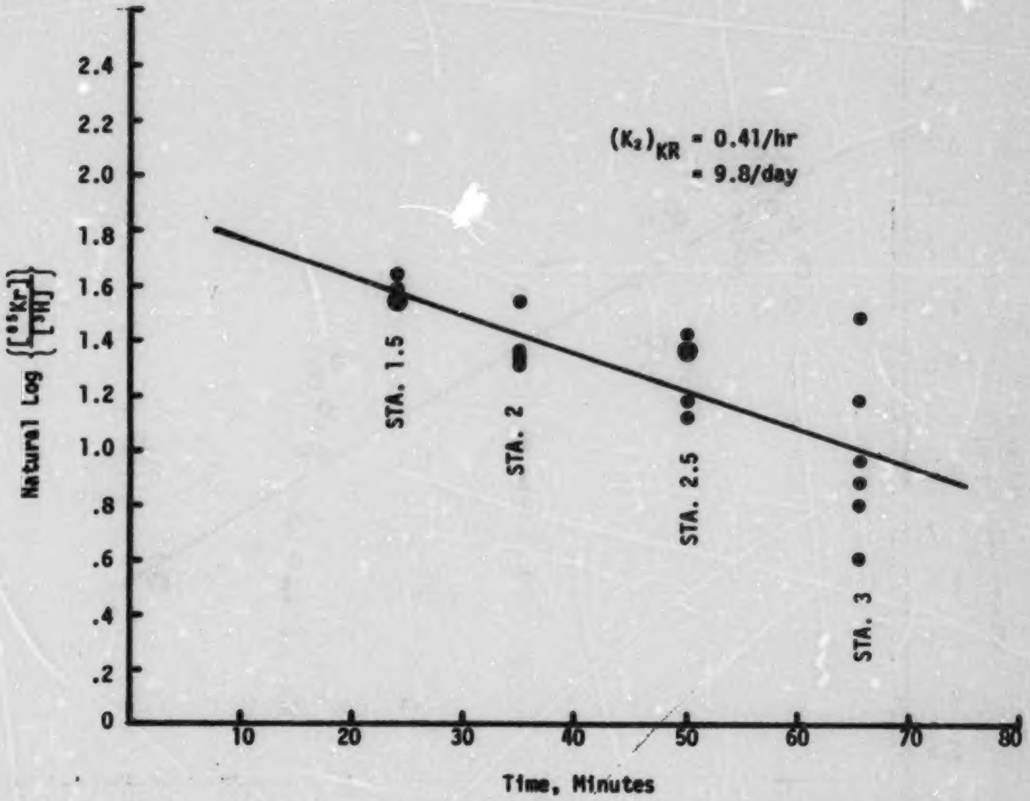


FIG. 9. Yaquina River Estuary Gas Transfer Data for Reaeration Project September 17, 1969

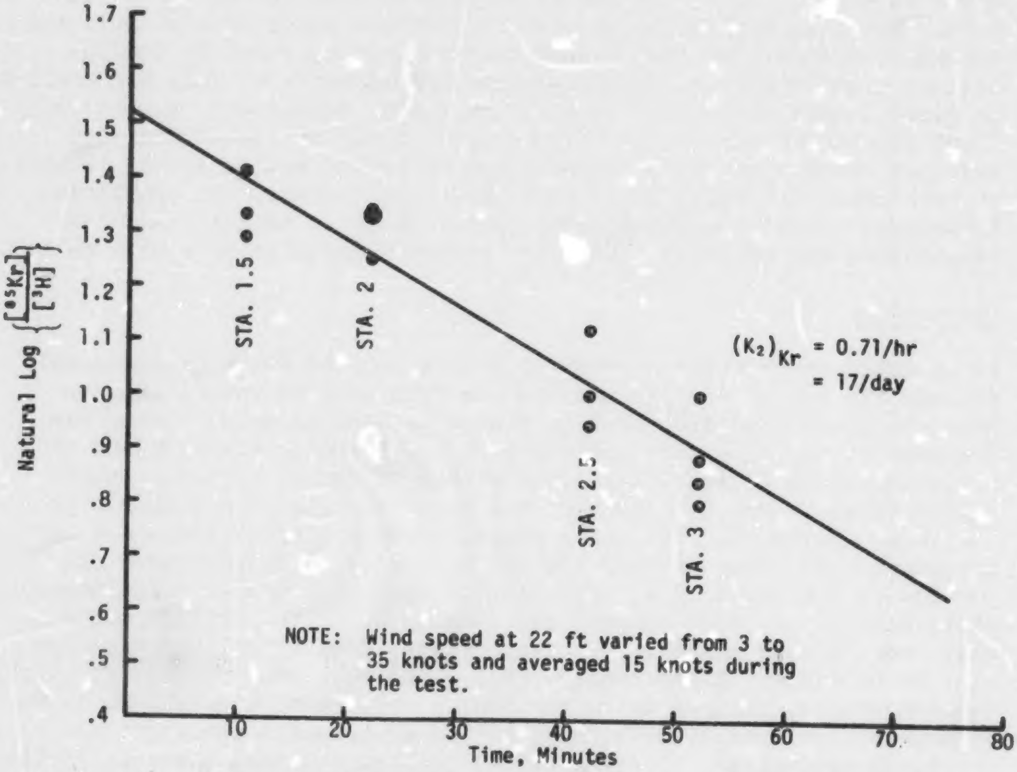


FIG. 10. Yaquina River Estuary Gas Transfer Data for Reaeration Project May 26, 1970

Equipment

Equipment, as our procedure, has been modified and supplemented as we progressed. Our work boat is equipped as shown schematically in Figure 11. A submersible pump is used to maintain positive pressure on all fittings above the water line. The only modification we plan is to add a second system for simultaneous sampling at a second depth. Our wind measurement system will consist of two Climet cup anemometers and direction vanes sampling at 8 and 26 feet above the water surface. Data are traced on two two-channel Esterline Angus recorders. As shown in Figure 12, the height is maintained constant by a floating dock. The dock is also to be used to indicate stage on a Leupold and Stevens recorder. Bottom channel currents are measured by Geodyne digital film recording, Savonius-type current meters. They are mounted on rigid tripod structures so that the speed sensors are respectively 1 and 6 feet off the bottom (Figure 13). A Marine Advisors ducted impeller current meter is buoy-mounted to record analog traces (Rustrak) of both speed and direction 1 foot below the surface. An Industrial Instruments induction salinometer (Model RS-5) is used to indicate temperature and salinity. Data are recorded manually by a crew member.

Discussion

In general, the krypton distribution in a body of water cannot model exactly the oxygen distribution in the same body of water; and, in general, the oxygen distribution cannot be used to model either gas transfer at the air-water interface or the vertical distribution of properties, due to insufficient knowledge of the biochemical interaction terms. Krypton, however, has been shown to be a linearly proportional analog for estimating oxygen transfer across the air-water interface. In order to estimate the influence of hydrodynamic properties on the vertical distribution of dissolved oxygen, simultaneous distributions of other properties, such as velocity, tritium water, salt, and in some cases either temperature, density or rhodamine dye must be provided. Historically, $(K_2)_{OX}$ has been used to predict oxygen concentrations, assumed to be vertically uniform to a depth, H, in an attempt to incorporate both the interfacial transfer and vertical transport mechanisms. Dobbins (1964) pointed out that there is little justification to inclusion of H in prediction formulas since H can be determined separately, thus removing its apparent influence on f, the gas transfer coefficient. H frequently has been interpreted as the total depth of the stream, although it should be obvious that since it is defined by $[V/A]$, in many ponds, estuaries, bays, fjords, oceans and possibly some rivers, it is related to a depth defined in terms of either boundaries or parameters indicating gradual or abrupt vertical flux inhibition. Variations in H may be greater or lesser than variations in total depth depending, among other things, on the presence and dynamic stability of pycnoclines. We reported our results today in terms of $[f(A/V)]_{KR}$ rather than f_{KR} because we have not

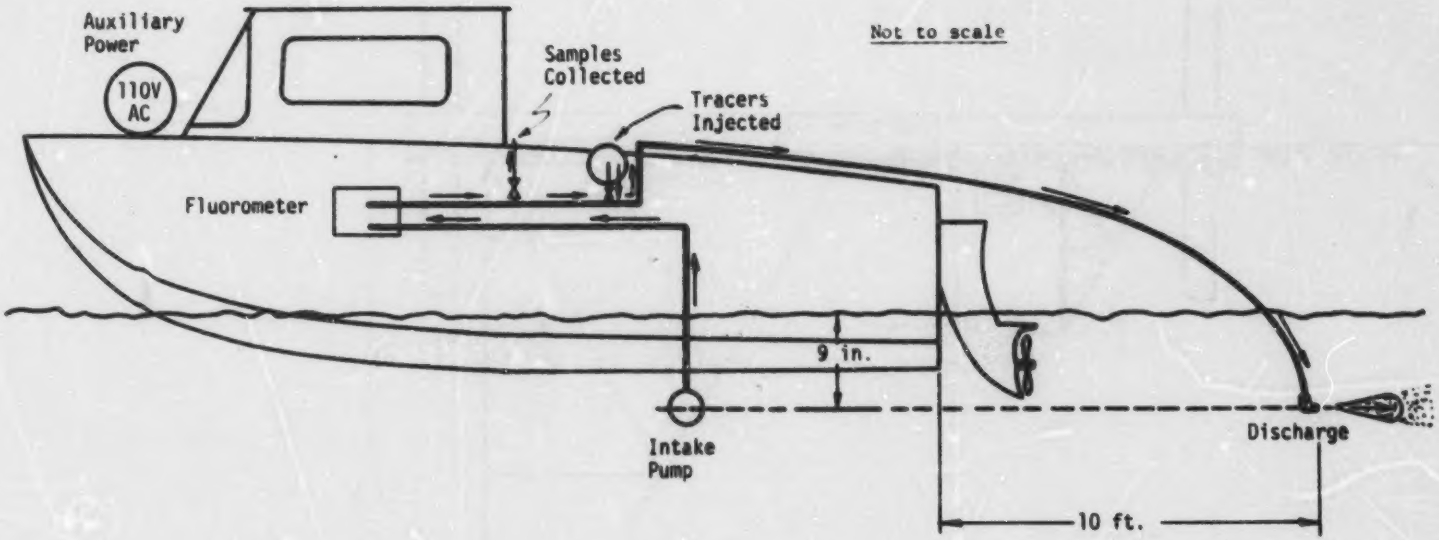


FIGURE 11. Research Vessel Adapted for Gas Transfer Data and Flourometry

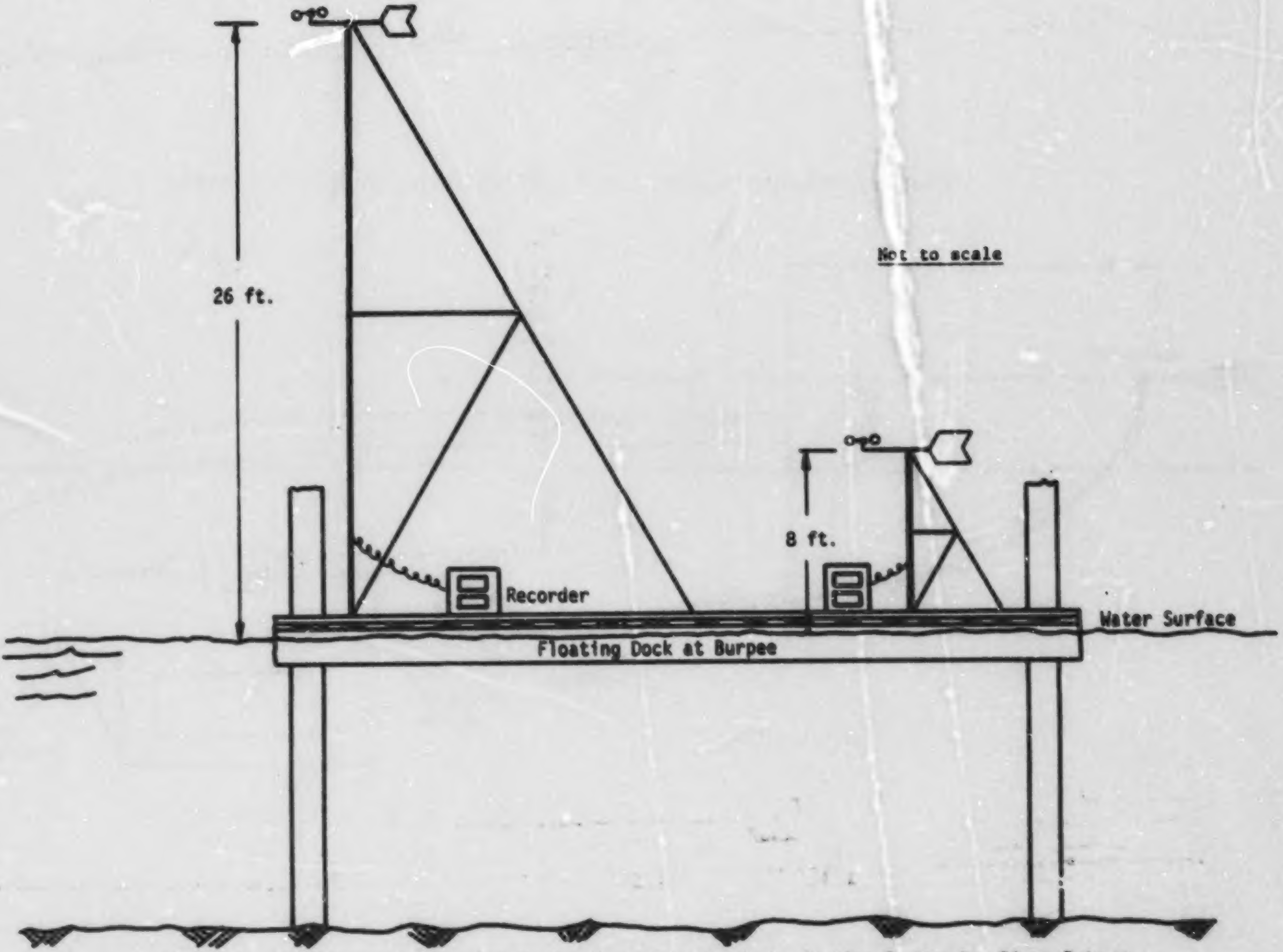


FIGURE 12. Wind Current Meter Set Up for Reaeration Project, Station 2, Yaquina River Estuary

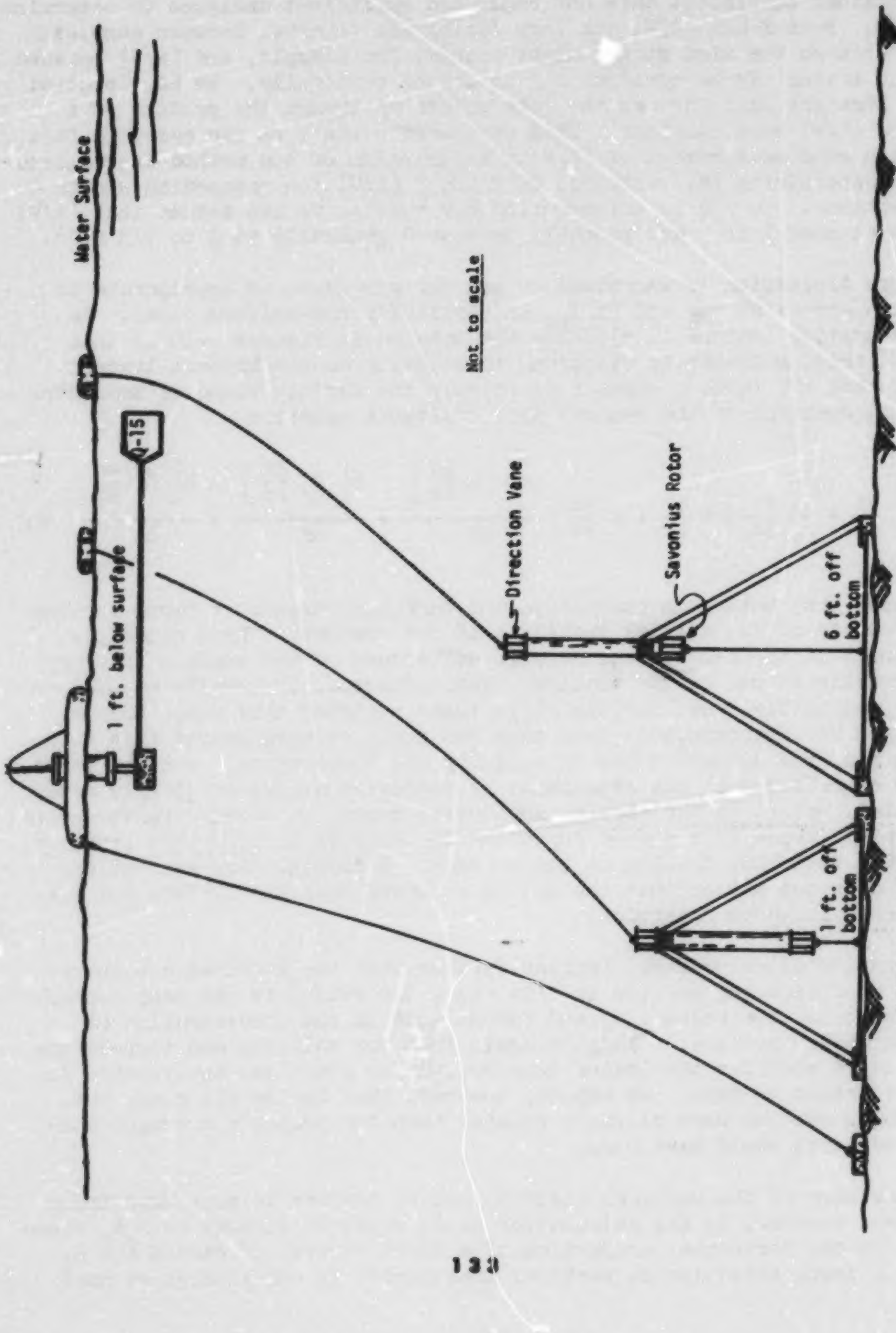


FIGURE 13. Water Current Meter Set Up for Reaeration Project, Yaquina River Estuary

obtained sufficient data nor conducted sufficient analyses to determine V/A. Both f and (A/V) can vary during the interval between samples; f because the wind stress might change, for example, and (A/V) because the tracer may be continuously dispersed vertically. We have plotted a straight line through the data points as though the product of f and (A/V) were constant. This of course needn't be the case, in fact, with good measurement of (A/V) a major value of the method is realized in determining the variation of f and $f\ (A/V)$ for succeeding stream sections. To aid in interpreting our results we can assume that (A/V) is at most 1 ft^{-1} and probably decreased gradually to $\frac{1}{2}$ to 1/3 ft^{-1}.

Some discussion is warranted on how our procedure is appropriate to measurement of f_{KR} and $(K_2)_{KR}$ in vertically non-uniform flows. An assumption implied in plotting the data as in Figures 7-10 is that the tritium tracer is dispersed identically as the krypton tracer so that the ratio changes reflect only the surface boundary transfer. An inspection of the general mass transport equation

$$\frac{\partial c}{\partial t} = -u\frac{\partial c}{\partial x} - v\frac{\partial c}{\partial y} - w\frac{\partial c}{\partial z} + \frac{\partial\left[D_x\frac{\partial c}{\partial x}\right]}{\partial x} + \frac{\partial\left[D_y\frac{\partial c}{\partial y}\right]}{\partial y} + \frac{\partial\left[D_z\frac{\partial c}{\partial z}\right]}{\partial z} + S.$$

shows that both mean convective and turbulent transport terms include products of the spatial gradients of the tracers. These generally cannot be the same in our experiment because of the surface boundary transfer of one of the tracers. The difference is usually assumed to be negligible, and, for the field tests reported here where tracer depth was approximately less than two feet, we have judged this to be so from the distributions of salinity and temperature. Caution is to be exercised when the experiment is conducted during or shortly after rains, which, in the Pacific Northwest, cannot be considered an unusual event. Aside from energy input and the effects on A, a thin lense of fresh water may develop on the surface. Accepting this assumption, it does not matter that the dye is released near the surface and disperses in three directions.

Equation 1, as written, implies further that the measured concentration of escaping gas (or in this case, the ratio) is the mean concentration of the volume, V, and furthermore is the concentration at the water "surface." Judging again from the salinity and temperature data we consider the latter true enough for practical application in our trials to date. We expect, however, that during all runs, the concentrations were slightly greater than the properly averaged concentration would have been.

Knowledge of the vertical distribution of tracers is much more important, however, to the calculation of V, which is defined as $H_e A_h$ where A_h is the horizontal projection of a short stretch of stream and H_e is a depth effective in vertical transport. In our studies we have not measured the vertical tracer distribution nor have we completed analysis of other data which would allow us to compute an estimated value for H_e. In some runs we expect H_e was increasing at a rate greater than in others.

It is important to recognize that H_e for the tracer experiment does not necessarily equal H_e for vertical transport of oxygen transferred at the surface, although it may approach it, and that H_e for oxygen would not necessarily be constant in time for an estuarine system. Under these circumstances it is clear that the best alternative is to release the tracer near the surface and measure, or compute, the vertical distribution of properties. To release the tracers at depth without knowing beforehand what the vertical density distribution is and what can be forecast for the ensuing tidal cycle is surely to invite trouble. If the tracer doesn't reach the surface, the surface transfer rate cannot be measured. To attempt uniform vertical distribution of tracer initially may require considerably more tracer (and dollars) than is necessary.

Using the procedure outlines, it takes two man days to set up meters, etc., and two men to conduct the test. It presently costs about $200 for the combined dose of tracers.

We intend to continue our experimental studies, adding measurement capabilities and increasing tracer doses as budget and other constraints allow.

Acknowledgements

Dr. Tsivoglou's assistance in the early stages of getting this project started and his continued interest and encouragement are gratefully appreciated.

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Radiological Safety

Jon P. Longtin

Consideration of the radiation safety aspects of the double tracer reaeration technique must be separated into two areas of concern:

1. Radiation exposure to the general population in the area of the reaeration study;
2. Radiation exposure to the personnel performing the study.

In order to gain approval from the Atomic Energy Commission (AEC) to release radioactivity to the environment, it is necessary to demonstrate that the proposed limits are not likely to cause any individual in the unrestricted environment to receive a dose to the whole body in any period of one calendar year in excess of 0.5 rem. This can most easily be done by using the nature of the dispersion of a point source of radioactivity in a flowing stream to show that MPC(w) levels are approached in a time less than that required for the material to reach a potable water intake. The MPC(w) values listed in 10CFR20, Appendix B, Table II, column 2, are such that continuous exposure at these levels to a specific radionuclide will not result in a dose to the whole body in excess of 0.5 rem per calendar year. For a mixture of several nuclides, the sum of the ratios of the concentration to the MPC(w)'s must be less than or equal to one in order to meet this dose condition.

In order to gain some perspective about the dispersion of material introduced as a point source, consider the one dimensional dispersion model for a conservative substance,

$$C(x,t) = \frac{M}{A \quad 4\pi D_L t} \exp \frac{(x-vt)^2}{4D_L} ,$$

and looking only at the peak concentration (x = vt),

$$C_p(t) = \frac{M}{A \quad 4\pi D_L t}$$

where

$C(x,t)$ = concentration at x at time t

M = mass of material dosed

A = cross sectional area of stream

D_L = dispersion coefficient

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t = time from dose

x = distance from dose point

v = stream velocity

According to Gloyna and Ledbetter (1), for flows ranging in size between small streams and large rivers, D_L equals about 0.02 miles^2/day. As an example, the following data, taken during a reaeration study from a reach on the Great Miami River between Dayton and Hamilton, Ohio, are used.

Table I

Station	Elapsed Time (hours)	H-3 (μCi/mlx10^{-5})	Kr-85 (μCi/mlx10^{-5})
Dose	0	1x10^8	0.5x10^8
1	2.41	5.97	1.25
2	4.58	2.75	0.430
3	8.99	0.597	0.083
4	12.66	0.394	0.062

Cross sectional area estimated at 1500 ft^2

Dose 1 Ci H-3

0.5 Ci Kr-85

Substituting parameters and making proper conversions yields

$$C_p(t) = 4.357 \times 10^{-5} \frac{1}{\sqrt{t}} \ \mu\text{Ci/ml}$$

for M = 1 curie. This curve is plotted along with the tritium data in Figure 1. It is seen that this model predicts a very rapid decrease in concentration in the first few minutes. In fact, solving for the time required for the tritium MPC(w) level to be reached (1x10^{-3} μCi/ml) yields about 7 seconds. Thus, in this example, almost from the very start it is to be expected that the tritium activity levels will be less than MPC(w).

At the end of one hour, the Kr-85 concentrations (for a half curie dose) is predicted to be 2.18x10^{-5} μCi/ml neglecting losses to the atmosphere. The difficulty in evaluating the significance of the concentration is that there is no MPC(w) given for Kr-85 and thus no direct standard exists to make a comparison. However, the primary point to consider is that the MPC(w) values can be averaged over a

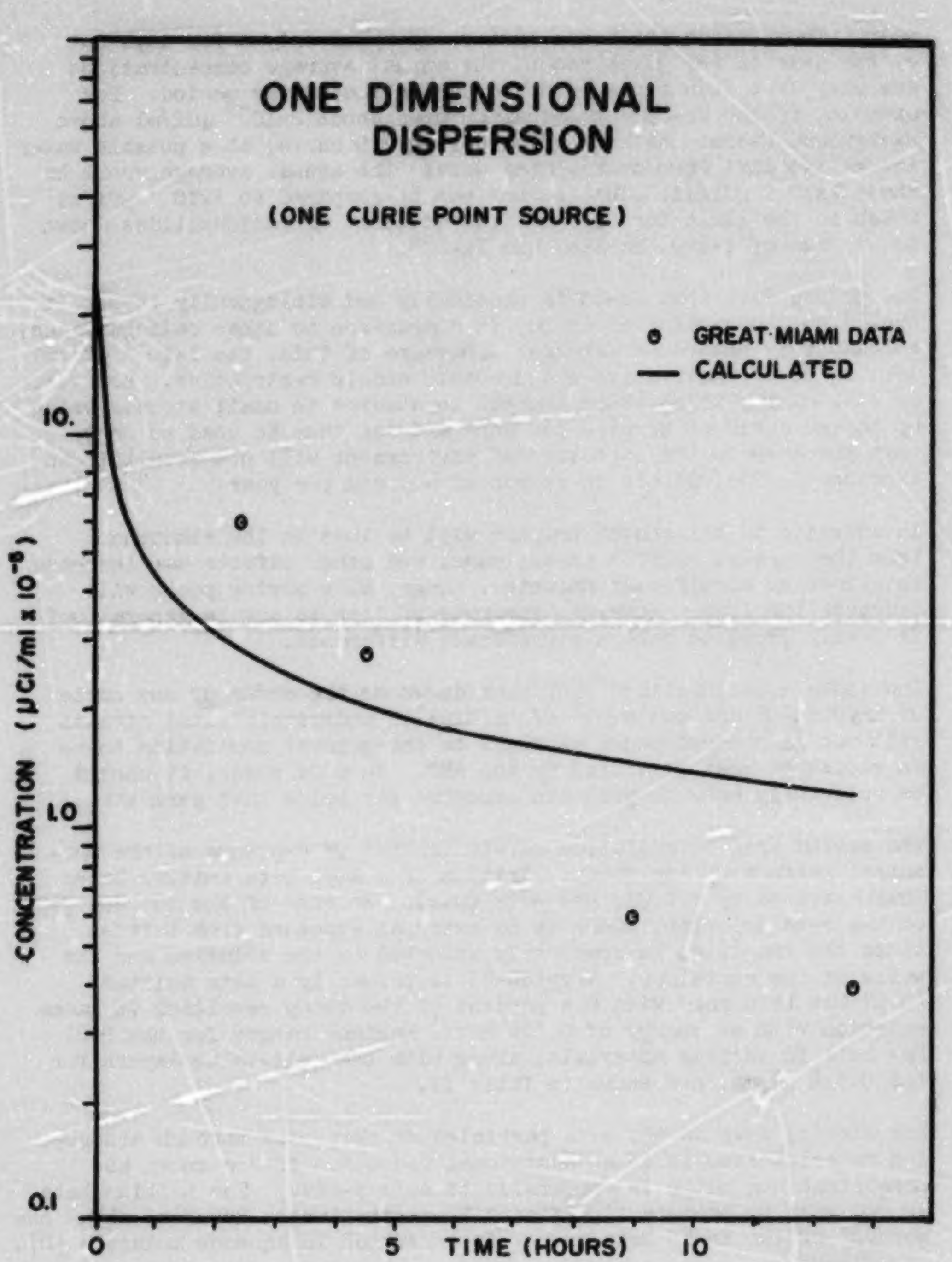
ONE DIMENSIONAL DISPERSION
(ONE CURIE POINT SOURCE)
GREAT MIAMI DATA
CALCULATED
CONCENTRATION (μCi/ml x 10⁻⁸)
10.
1.0
0.1
0
5
TIME (HOURS)
10

FIGURE 1

year. Thus, since the reaeration study spans only a few days out of the year in any given reach, the annual average concentrations are much less than those produced during the survey period. For example, if the Kr-85 concentration were about 2×10^{-5} μCi/ml above background (about one hour after dosing 0.5 curie) at a potable water intake two days (two overlapping doses) the annual average would be about 1×10^{-7} μCi/ml. This number can be compared to 1×10^{-7} μCi/ml which is the limit for unidentified mixtures of radionuclides known to be free of I-129, Ra-226, and Ra-228.

Due to the fact that Kr-85 is chemically and biologically inert, it should not be considered to be, in comparison to other radionuclides, a relatively hazardous material. Because of this, the 1×10^{-7} μCi/ml unit is very conservative and probably unduly restrictive. However, it is a number which is approached in studies in small streams using up to one curie of krypton per dose and can thus be used to show that the dose in the unrestricted environment will not result in an exposure to individuals in excess of 0.5 rem per year.

In addition to the above, krypton will be lost to the atmosphere from the stream. Riffle areas, dams, and other effects can increase this loss by significant amounts. Large, slow-moving pools will decrease the loss. However, the rate of loss is not in general sufficiently great to make a significant difference.

The above considerations show that doses on the order of one curie of krypton-85 and one curie of tritium in moderately sized streams will not in general cause exposure to the general population to be in excess of that permitted by the AEC. In most cases, it should be relatively easy to maintain exposure far below that permitted.

The second area of radiation safety is that of exposure of the personnel performing the study. Tritium is a soft beta emitter being characterized by a 0.019 Mev beta (max). Because of the low energy of the beta emission, there is no external exposure from tritium since the radiation is completely absorbed by the solution and the walls of the container. Krypton-85 is primarily a beta emitter (0.67 Mev beta max) with 0.4 percent of the decay resulting in gamma emission with an energy of 0.514 Mev. Maximum ranges for the 0.67 Mev beta in various materials, along with the half-value layers for the 0.514 gamma, are shown in Table II.

The slowing down of the beta particles as they pass through absorbing material results in an additional radiation source known as Bremsstrahlung which is comparable to soft x-rays. The tritium beta is too weak to produce significant Bremsstrahlung. However, about one percent of the Kr-85 beta energy is converted in aqueous solution (2).

TABLE II

Material	Max range (inches)	Half value layer
Water	0.09	7.8 inches
Glass	0.04	-
Air	70	236 feet
Lead	0.007	4.1 millimeters

Thus, the bulk of the external exposure from dose solutions containing tritium and krypton-85 will be from the gamma radiation and from the Bremsstrahlung associated with the krypton beta. An estimate of the dose from a point source of one curie of krypton-85 can be made from the following expressions taken from the Radiological Health Handbook (3):

$$\text{Dose (mr/hr)} = \frac{NI\gamma}{S^2}$$

$$I\gamma = 0.156\ n\ E\ (10^5\ \mu a)$$

where: N = number of millicuries
S = distance to source (meters)
n = gamma quanta per disintegration
E = gamma energy in Mev
μa = air absorption coefficient (cm^{-1})

For $N = 1000$ mCi, $n = 0.004$, $E = 0.514$, and $\mu a = 3.87 \times 10^{-5}$

Dose (mr/hr/curie at 1 meter) = 1.24

Measurements from liter bottles of dose solution containing one curie of Kr-85 at 18 inches gave dose readings of about 8 mr/hr. This can be compared to a calculated value of 5.9 mr/hr at 18 inches.

Tritium can enter the body by inhalation exchange in the lungs and absorption directly through the skin. Trace levels of tritium have been observed in the urine of individuals performing the dosing operation. There are two points at which there is potential for direct exposure to the radioactive tracers and thus internal dose. The first is the sampling or assaying of the dose solution. During the removal of a sample of solution via syringe, krypton gas and tritiated water vapor are apt to be released to the atmosphere. If the procedure is performed in the open air, there is little build-up in the immediate area and consequently little potential for exposure. If, on the other hand, the procedure is done in a confined area, it is possible to detect temporary increases in background as krypton escapes. Thus, when working in a confined area, a hood, if available, should be used.

In the absence of a hood, fans, open windows, etc., should be employed. In addition, the area should be monitored with a survey meter and the occupancy controlled until normal background is reached.

The second direct exposure can occur during dosing in the stream. If the actual dosing is done manually, that is, breaking the dose container by hand in the stream, there is a temporary potential exposure to krypton gas and tritiated water vapor in the air and an immersion in fairly concentrated solution in the eddy caused by the water flowing around the dosing personnel. Wearing rubber waders which do not leak and keeping exposed areas of the body from contacting the water immediately after dosing will minimize exposure.

Occupational exposures recommended by the NCRP and required by 10CFR20 for licensees of by-product material limit the accumulated whole body dose to a maximum of 5 (N-18) rems where N equals the age of the individual in years. In addition, a limit of 3 rems per calendar quarter (13 weeks) is imposed.

The dosing procedure requires only a few minutes of close exposure to the dosing solution. Thus, it can be expected that the whole body radiation dose per application of a one curie Kr-85 solution will be on the order of one mr. The highest dose rates occur to the hand when handling the dose container. Calculating the dose rate for one curie of Kr-85 at the surface of a one liter container (5 cm) results in about 500 mr/hr. Such handling requires about one minute and results in a dose of about 8 mr to the hands.

The following procedures are recommended to minimize radiation exposure to personnel:

1. Use lead shielding wherever and whenever possible. Carry out dose preparation and assaying behind lead bricks.
2. During storage and transporting, maintain adequate distances from source material and personnel. If possible, do not concentrate all the source material required for a study in one location unless adequate shielding is available.
3. Vary personnel assignments - involve as many individuals as possible. Use different groups of individuals for dose preparation and dose release, thus distributing the dose and minimizing individual exposure.

For minimizing exposure to the general population:

1. Use only the quantities of tracers that are absolutely required. Excessive quantities are not only costly, but inconsistent with a policy of minimizing the release of radioactivity to man's environment.
2. Plan the study and release points such that the tracer concentrations will be essentially undetectable at points of water use.

Monitoring of the personnel is required by by-product licensees. Film badges are routinely used for this purpose in addition to pocket dosimeters and survey meters (which record external exposure in mr/hr). Because of the potential escape of krypton when preparing or assaying dose solutions, it is advisable, when working in a confined space, to have available a direct reading survey meter. It is preferable to use one with an audible alarm. To date, film badge readings during reaeration studies have not shown exposures above background.

Water use points such as water intakes should be monitored by obtaining water samples during critical periods of the study.

In summary, while the total quantity of radioactivity involved in the reaeration measurement technique can be fairly large, the rapid dispersion in the stream coupled with the low relative hazards associated with tritiated water and krypton-85 insure that, with proper planning and execution, the exposures to the general population and to the personnel conducting the study can be kept well below those permitted by the AEC.

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Effect of Hydraulic Properties on Reaeration

Edward L. Thackston

Introduction

The ultimate objective of the current studies to measure reaeration rates under many different conditions is to improve our ability to predict the reaeration coefficient from basic hydraulic data. The measurement of the reaeration coefficient by the tracer method is accurate and reliable, but it is also expensive and time-consuming, and it requires highly-skilled personnel and a major investment in instrumentation. Most organizations interested in obtaining reaeration coefficients cannot justify this investment.

However, many times basic hydraulic data (discharge, depth, slope, velocity, temperature) are available for the reach of stream in question, or they can be obtained for less expenditure than a tracer test would require. It is quite possible that acceptably accurate average values of the hydraulic parameters could be obtained with significantly fewer measurements than the research program described earlier has used. One of the valuable conclusions to result from this research will be an indication of how extensive a stream survey is necessary in order to obtain values of the hydraulic parameters which are sufficiently close to the true average values to give "acceptable" results when used in a mathematical model for prediction of the reaeration coefficient.

Previous Attempts at Reaeration Prediction

There have been many previous attempts to relate the reaeration coefficient to hydraulic variables, so it could be predicted from a knowledge of the hydraulic data from a particular stream. The prediction equations discussed in this section are those which have been proposed for actual field use. There have been many other reaeration equations derived to fit the data from a particular set of laboratory tests, usually by empirical correlation techniques, but those equations have not found use by engineers in practice.

Black and Phelps - The first attempt to predict oxygen transfer into polluted water was a method developed by Black and Phelps(2) in 1910 for prediction of reaeration in the New York harbor. The prediction equation assumed that reaeration was accomplished by molecular diffusion alone and was based on Stephan's(17) solution to Fick's second law of diffusion. The equation predicted the average oxygen concentration in a vertical column of quiescent water at a given time after being exposed to the atmosphere.

Velz(24) applied this concept to flowing streams in 1938. He assumed reaeration by molecular diffusion under quiescent conditions for given periods interspersed by periodic instantaneous complete mixes which destroyed the vertical oxygen profile and mixed the oxygen which had been absorbed at the surface downward into the entire depth of stream. The time between mixes was empirically correlated with stream depth and velocity. Although not physically realistic, the method is still used by some workers and may give reasonable results because of the form of the empirical correlations developed for the ficticious "time of mix".

Streeter and Phelps - The concept of a "reaeration coefficient", k_2, was first proposed by Streeter and Phelps(18) in 1925 in their report of studies of the pollution of the Ohio River which produced the first equation for the oxygen sag curve. They suggested that

$$k_2 = C\frac{U^n}{H^2} \qquad (1)$$

in which C and n are "constants", U is the average velocity, and H is the river stage above extreme low water.

Streeter and Phelps found that, on the Ohio River, k_2H^2 was proportional to U^n, but values of C ranged from 0.23 to 130.0, and values of n ranged from 0.57 to 5.40 for the different reaches. They showed graphically that there was a relationship between C and low water slope and channel irregularity, a crude measure of roughness. They also graphically related n to the relationship between velocity and river stage primarily a measure of channel shape.

The results demonstrated that a simple relationship between k_2 and average velocity and depth is unrealistic. Other factors, such as roughness, channel slope, and channel shape, must also be considered when U and H are the primary variables. Although Streeter and Phelps were the first to attempt a prediction of k_2, their work demonstrated a more thorough insight into the factors which influence reaeration than many of the investigators which followed.

O'Connor and Dobbins - The first rational attempt to predict reaeration coefficients from basic hydraulic data was by O'Connor and Dobbins(14), using a hypothesis based on an oxygen transfer model developed by Dobbins(6). Dobbins considered that the controlling factor in oxygen absorption was the resistance of a liquid film at the surface, through which oxygen must move by molecular diffusion. The film was assumed to be constantly renewed by unsaturated elements from the body of the stream through the mechanism of turbulence, as originally proposed by Higbie(9). The rate of transfer through an element of the surface depends on the length of time it has been exposed to the atmosphere. The function describing the age distribution of surface elements was taken to be that of Danckwerts(4), which is

$$\int_{t}^{t+dt} f(t)dt = re^{-rt} \tag{2}$$

in which f(t)dt = the relative part of the surface area having ages between t and t + dt, and r = the rate of surface renewal. By combining Equation 2 with Fick's law of diffusion,

$$\frac{\partial m}{\partial t} = - D_m A \frac{\partial c}{\partial y} \tag{3}$$

in which $\partial m/\partial t$ = the rate of mass transfer, D_m = the molecular diffusion coefficient, and A = the surface area, Dobbins obtained the relation

$$K_L = (D_m r)^{\frac{1}{2}} \coth(\frac{rL^2}{D_m}) \tag{4}$$

in which K_L = the liquid film coefficient, and L = the thickness of the surface film. Because the coth term is very close to unity in the normal streamflow range and $K_L = K_2/h$, O'Connor and Dobbins concluded that

$$k_2 = \frac{(D_m r)^{\frac{1}{2}}}{2.30h} \tag{5}$$

The rate of surface renewal was considered to be the ratio of the vertical rms velocity fluctuation at the surface, $\sqrt{\overline{v'^2}}$, to the mixing length at the surface, l_m. Because, according to the Prandtl mixing length theory,

$$\sqrt{\overline{v'^2}} = l_m \frac{du}{dy} \tag{6}$$

r may be expressed as

$$r = \frac{du}{dy} \qquad (7)$$

In developing the formula for isotropic turbulence, the measurements of Kalinski(11) in the Mississippi River were used. These measurements showed the vertical velocity fluctuation to be about 10% of the mean velocity and the mixing length to be about 10% of the mean depth. Thus,

$$r = \frac{\sqrt{\overline{v'^2}}}{l_m} = \frac{0.1U}{0.1h} = \frac{U}{h} \qquad (8)$$

Substitution of Equation 8 in Equation 5 yields

$$k_2 = \frac{\sqrt{D_m U}}{2.30\ h^{3/2}} \qquad (9)$$

The results of a laboratory investigation were presented to show that k_2 was proportional to $r^{1/2}$, and a collection of "measured" and predicted values of k_2 were given which apparently demonstrated close correlation.

Churchill, Elmore, and Buckingham - Churchill, Elmore, and Buckingham(3) presented an analysis of 30 measurements of k_2 in streams in the Tennessee Valley. They are believed to be the most reliable group of field scale reaeration data reported in the literature. These data were measured on unpolluted streams under conditions of uniform steady flow, and all the effects of photosynthetic activity were systematically eliminated. Churchill, et al., used the methods of statistical correlation analysis to derive as a formula for the prediction of k_2 at 20°C,

$$k_2 = 5.026\ U^{0.969}\ h^{-1.673} \qquad (10)$$

They also concluded that the inclusion of several dimensionless groups of hydraulic variables would not significantly improve the fit of the equation to the observed data, as long as velocity and depth were used. However, no other single variables were tried in combination with depth and velocity or instead of one of the primary variables.

Dobbins - In 1964, Dobbins(7) proposed new equations, based on the Kolmogoroff similarity principle, for the determination of r and L in Equation 4, which were

$$L = C_4 \left(\frac{v^3}{E}\right)^{\frac{1}{4}} \tag{11}$$

and

$$r = \frac{C_5}{C_4} \frac{\rho \upsilon^{3/4} E^{3/4}}{\sigma} \tag{12}$$

in which υ = the kinematic viscosity, σ = the surface tension, ρ = the density, and C_4 and C_5 = constants. The factor C_5 should be an absolute constant, while C_4 should be determined by the dynamics of the system. Through a series of laboratory experiments, C_5 was determined to be about 14.3.

Dobbins(8) later combined these equations, expressed υ, σ, ρ, and D_m as functions of temperature, and proposed for the prediction of k_2,

$$k_2 = \frac{0.12\ C_A A E^{3/8} \coth\left(\frac{B E^{1/8}}{C_4^{\frac{1}{2}}}\right)}{C_4^{3/2} h} \tag{13}$$

in which $C_4 = 0.9 + F$; $C_A = 1.0 + F^2$; $A = 9.68 + 0.054(T - 20)$; $B = 0.976 + 0.0137(30 - T)^{3/2}$; $E = 30.0\ SU$; and $F = U/\sqrt{gh}$. The term C_A = the ratio between the actual interfacial area and the projected surface area; F = the Froude number; S = the slope in ft per 1000 ft; U = the velocity in ft per sec; T = the temperature in C; and k_2 is given as per day. The constant C_4 was evaluated using the data of O'Connor and Dobbins(14), Churchill, et al.(3) and Krenkel(12).

Both the original formulas of O'Connor and Dobbins(14) and the newer formulas of Dobbins(7) are based on the relation which had been derived earlier by Dobbins(6)(Equation 4). However, the basic premise that r is equal to the velocity gradient at the surface, as well as the methods chosen to express it, has been questioned(16, 5).

The expression for the rate of surface renewal in the isotropic flow formula, while possibly less rational than the method used in the now-abandoned non-isotropic flow equation, is apparently more nearly correct. The measurements in the Mississippi River reported by Kalinske(11) showed the velocity fluctuation to be about one-tenth the mean velocity. This produced the simple relation shown by Equation 8, which was assumed to hold universally. However, the data of Kalinske clearly showed that $\sqrt{\overline{u'^2}}/U$ was a function of the distance from the bottom. For the six different measurement series, the depth at which $\sqrt{\overline{u'^2}}/U = 0.1$ varied from about 0.27 of the total depth to about 0.55 of the total depth. Most measurements showed values of 0.1 at about 0.4 of the total depth and values of 0.08 near the surface. The total depth averaged about 19 ft. If $\sqrt{\overline{u'^2}}/U$ is a function of the distance from a solid boundary, a value of 0.1 should occur at the surface when the depth is about six ft to ten ft. This is precisely the range of depths for which the isotropic flow formula fit the data of Churchill best. At shallower depths, where the Kalinske data would predict a value of $\sqrt{\overline{u'^2}}/U$ of 0.12 to 0.20, instead of 0.1, the predicted values were too low. If $\sqrt{\overline{u'^2}}/U$ had been assumed to be 0.15 for these shallower reaches in accordance with the data of Kalinske, the agreement would have been much better.

An opposite effect, possibly caused by variations in mixing length rather than the velocity fluctuation, takes place at very shallow depths. The isotropic flow formula overestimates values of k_2 measured in a laboratory flume by a factor of 10.

The reason cannot be due to dropping the hyperbolic cotangent (coth) term, because any deviation of coth $(rL^2/D_m)^{\frac{1}{2}}$ from unity must increase k_2. Thus, ignoring the coth term would make the predicted value of k_2 too small. This is not in accord with the actual effect.

The somewhat questionable method of estimating the rate of surface renewal for this formula probably accounts for its inability to accurately cover a wide range of depth scales. Another possible reason may be in the assumption that k_2 is proportional to the square root of the rate of surface renewal. Diachishin(5) demonstrated that the experimental verification of this assumption in the laboratory was insufficient because the data also supported the hypothesis that k_2 was proportional to r.

The formulas of Dobbins(7) were the result of an attempt to develop more rational expressions for the rate of surface renewal and the thickness of the surface film. The proposed relationships were based on logical concepts, but attempts to establish numerical values for the hypothesized constants and proportionality factors were somewhat less than successful. In particular, the variation in the "observed" values of C_4, the ratio of surface film thickness to minimum eddy size, did not support the proposed relationship and was criticized by Thackston and Krenkel (19). The failure of this proposed proportionality to be verified casts serious doubt on the validity of all the relationships, since they all depend on C_4.

The approach of Churchill(3) differed radically from that of O'Connor and Dobbins. His equation was based on a correlation analysis of observed values of k_2 and various hydraulic parameters. The resulting equation naturally fit his particular group of experimental data better than any other, but it cannot be considered reliable outside the range of hydraulic variables used in its derivation. There is also some evidence that, on occasion, it does not give reasonable answers when the values of the hydraulic variables are within the original range.

Thackston and Krenkel - Thackston and Krenkel(20, 23) recently proposed a new approach to reaeration coefficient prediction, based on the vertical mixing coefficient. The basic premise was that k_2 was proportional to some function of surface turbulence or renewal and inversely proportional to some function of depth, representing both the water volume and the resistance to complete vertical mixing. The measure of surface turbulence chosen was the vertical mixing coefficient at the surface. The vertical mixing coefficient was assumed to be equal to the momentum transfer coefficient, which can be expressed as

$$\varepsilon_y = \kappa u_* y \left(1 - \frac{y}{h}\right) \qquad (14)$$

in which κ = the von Karman coefficient and y is the distance from the surface. Equation 14 predicts the value of ε_y at the surface to be zero. Experimental observations(10) indicate, however, that ε_y does have a small but finite value at the surface.

An expression for the average value of ε_y may be obtained by integrating Equation 14 with respect to y and dividing by the total depth. The resulting equation is

$$\overline{\varepsilon_y} = \frac{\kappa}{6} h u_* \qquad (15)$$

Solving Equation 15 for u_*, and substituting in Equation 14,

$$\varepsilon_y = 6\, \overline{\varepsilon_y}\, \frac{y}{h} \left(1 - \frac{y}{h}\right) \qquad (16)$$

Equation 16 indicates that the value of ε_y at any relative depth, y/h, is linearly proportional to $\overline{\varepsilon_y}$, which, from the work of Al-Saffar(1), may be taken as $\kappa h u_*/6$. Thus, it may be assumed that k_y at any relative depth, including the surface, is proportional to hu_*, or

$$k_{y_{surface}} = C_1 \frac{\kappa}{6} hu_* = C_2\ hu_* \qquad (17)$$

Thackston and Krenkel (22) have shown that, in uniform two-dimensional flow, D_L can be expressed as

$$D_L = C_3\ hu_* \qquad (18)$$

where D_L is the longitudinal mixing coefficient. Equation 18 has the same form as Equation 17, indicating that, in uniform two-dimensional flow, k_y at the surface is proportional to D_L. Thus the relationship shown by Krenkel and Orlob (13) between k_2 and D_L in uniform, two-dimensional flow must also hold for k_y, and, therefore,

$$k_2 = C_4 \frac{k_{y_{surface}}}{h^2} \qquad (19)$$

Substituting Equation 17 into Equation 19 and simplifying,

$$k_2 = C_4 \frac{C_2 hu_*}{h^2} = C_5 \frac{u_*}{h} \qquad (20)$$

Non-linear least squares regression analysis (21) using laboratory data, the data of Churchill, et al. (3), and the data of O'Connor and Dobbins (14), showed that, in each case, the best fit exponents to u_* and h were close to 1 and -1, as predicted. The constants for the three groups of data were 0.000215, 0.000208, and 0.000171, respectively.

The variable coefficient was assumed to be at least partially attributable to the increase in surface area at high Froude numbers, and a form of the constant C_A was introduced, making the final version of the equation

$$k_2 = 0.000125(1 + F^{\frac{1}{2}}) \frac{u_*}{h} \qquad (21)$$

or, reduced to fundamental hydraulic variables,

$$k_2 = 0.000125\left[1 + \left(\frac{U}{\sqrt{gh}}\right)^{\frac{1}{2}}\right]\sqrt{\frac{S_e g}{h}} \qquad (22)$$

The formula was tested on the field data of Churchill (3) and O'Connor and Dobbins (14), along with 3 other prediction formulas, the the results are shown in Table 1.

TABLE 1 - STANDARD DEVIATION OF PREDICTION BY DIFFERENT PREDICTION FORMULAS

Formula	Standard Deviation (1/day)
Thackston-Krenkel	0.369
Churchill, et al.	0.383
Dobbins	0.425
O'Connor-Dobbins	0.443

The Thackston-Krenkel equation was thus shown to be slightly more accurate than any of the others based on the limited field data available. The inclusion of the Churchill equation, which apparently demonstrates good correlation, in the comparison is really not valid. The Churchill equation was developed explicitly to fit 30 of the 64 observations used in the comparison, and thus possesses a significant positive bias. Computation of the value of F between the Thackston-Krenkel equation and the Dobbins equation yields a value of 1.33, implying statistical significance at approximately the 13% level.

Plots of observed and predicted values of k_2 are shown in Figures 1 through 4 for the four equations listed in Table 1. An examination of these figures will disclose the regions in which each equation is accurate or inaccurate, and the degree to which the points cluster about the 45° line is an indication of the overall predictive ability of the equation. The figures also disclose that some data, such as that of Owens, Edwards, and Gibbs (15), cannot be predicted by any of the equations discussed.

Summary of Hydraulic Variable Effects

The approximate effect of the different hydraulic variables on k_2, as predicted by the various equations, can be deduced from Table 2, which tabulates the exponent to which each variable is raised in each equation. In some equations, a particular variable is not raised to a simple constant power, so a complete comparison is not possible.

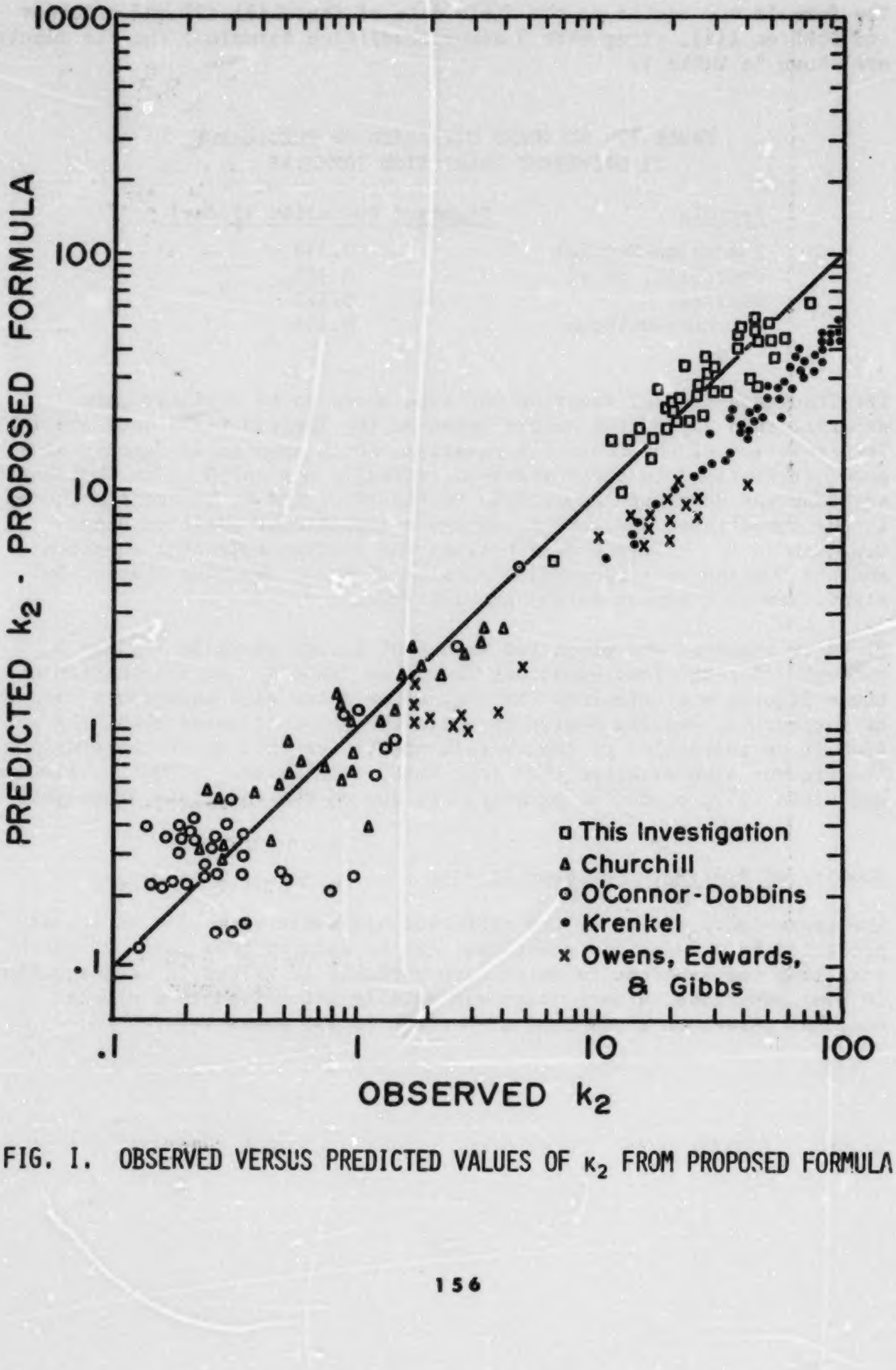

FIG. I. OBSERVED VERSUS PREDICTED VALUES OF κ_2 FROM PROPOSED FORMULA

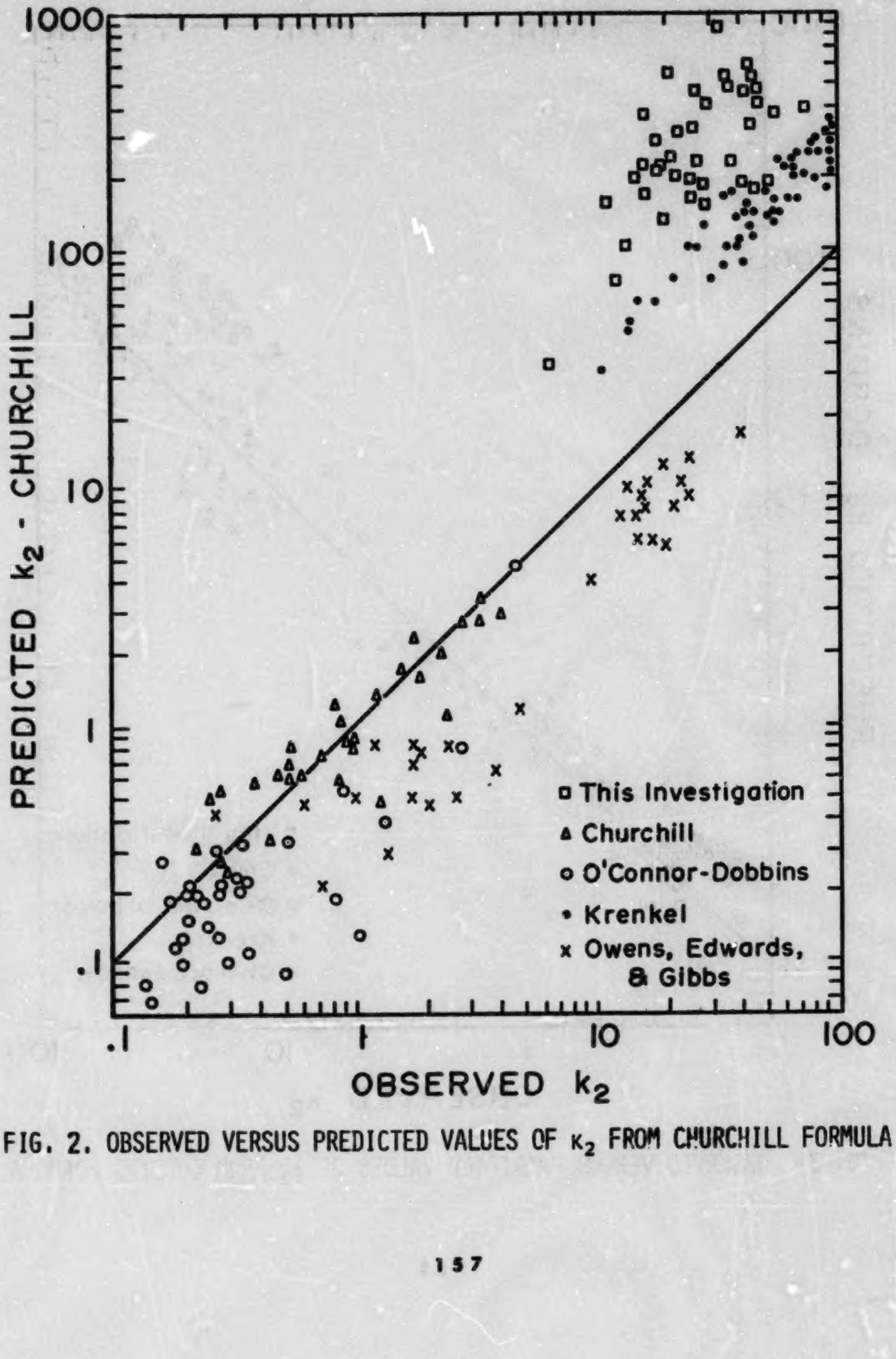

FIG. 2. OBSERVED VERSUS PREDICTED VALUES OF κ_2 FROM CHURCHILL FORMULA

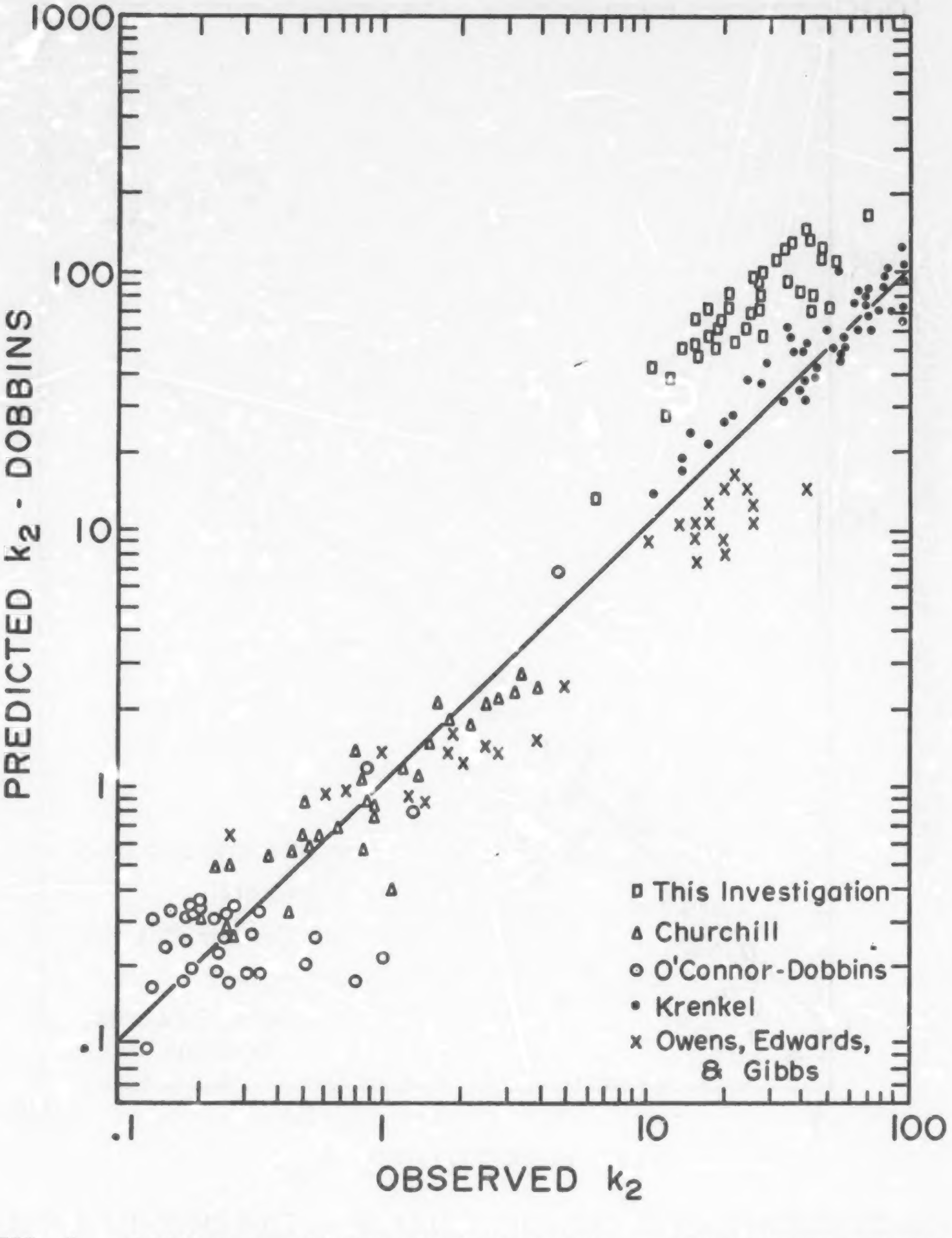

FIG. 3. OBSERVED VERSUS PREDICTED VALUES OF κ_2 FROM DOBBINS FORMULA

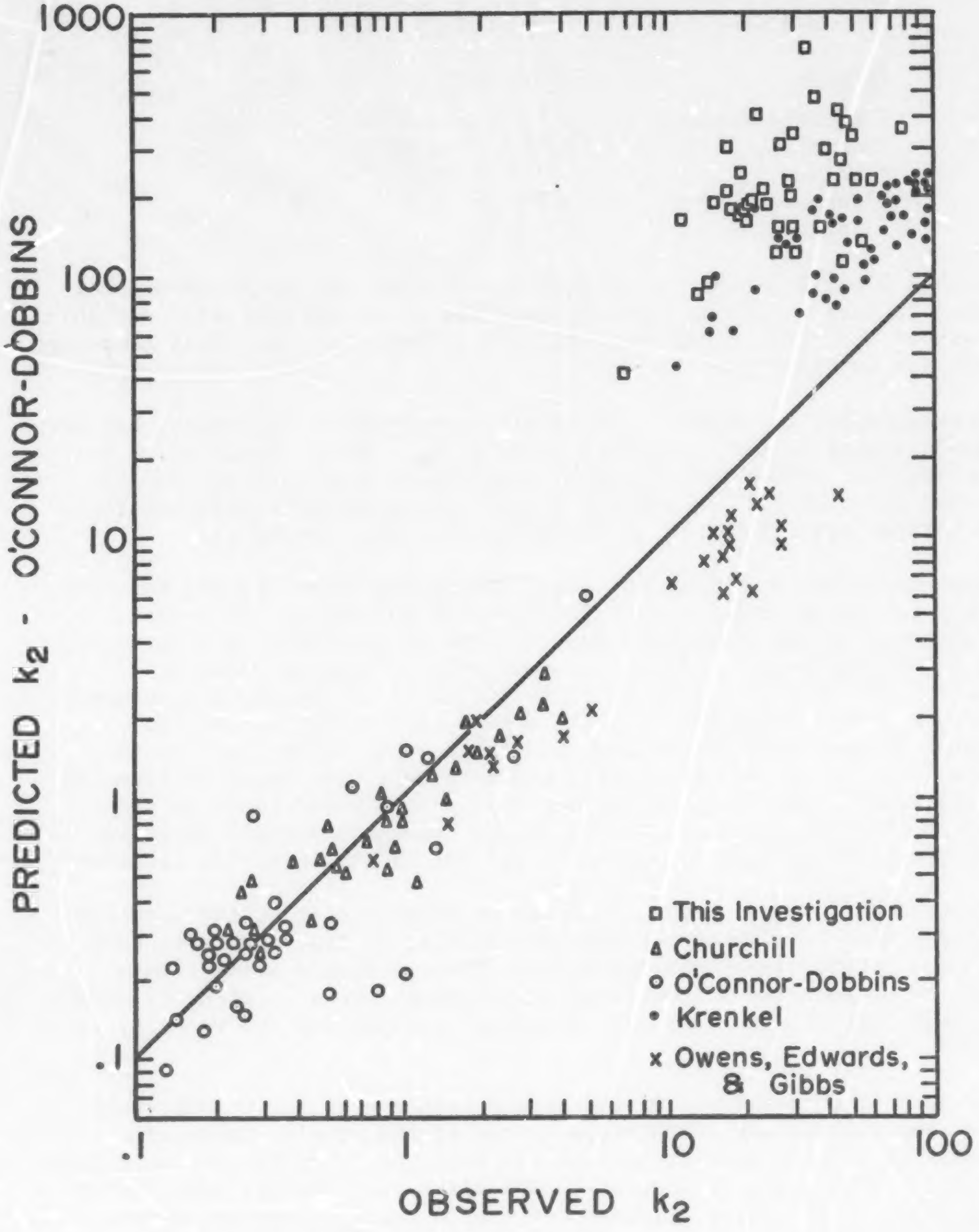

FIG. 4. OBSERVED VERSUS PREDICTED VALUES OF κ_2 FROM O'CONNOR-DOBBINS FORMULA

TABLE 2 - POWER TO WHICH EACH HYDRAULIC VARIABLE IS RAISED IN VARIOUS REAERATION FORMULAS

Formula	U	h	S_e
Thackston-Krenkel	~0.1	~-0.5	0.5
Churchill, et al.	0.97	-1.67	0
Dobbins	~1.0	~-1.5	~0.37
O'Connor-Dobbins	0.5	-1.5	0

From Table 2 and Equation 22, it can be seen that the Thackston-Krenkel equation is less sensitive to changes or errors in the hydraulic variables than any of the other equations. This is an important practical advantage in addition to its better accuracy.

The average velocity, which is difficult to determine accurately, has very little influence on the calculated value of k_2. Even a large error in estimating the average velocity would cause only a small error in the predicted value of k_2. On the other hand, an error in velocity would cause a large error in k_2 as estimated by the other equations.

The most important practical advantage of the Thackston-Krenkel equation, however, is the relatively small influence of the depth. It appears approximately to the one-half power, whereas it appears to approximately the three-halves power in the other equations. Thus, an error in estimating the average depth, which is difficult to determine accurately, will cause a much lower error in the values of k_2 predicted by the Thackston-Krenkel equation than by the three other equations. As an example, consider a case in which a stream with a true depth of five feet was estimated to have a depth of four feet. This would cause an error of 10% in the value of k_2 predicted by the Thackston-Krenkel equation, 27% by the O'Connor-Dobbins equation, and 30% by the Churchill equation.

The cost of the stream surveys is directly proportional to the accuracy required. Since the Thackston-Krenkel equation is less sensitive than the others to data error and inaccuracy, its use should allow stream surveys to be made for a lower cost, since fewer cross sections or depth meaurements are required and only a general estimate of the velocity is required.

Its dependence on the slope of the stream rather than the velocity also should simplify its use. The slope is one of the simplest hydraulic variables to measure accurately, and can sometimes even be estimated from USGS maps with acceptable accuracy. The slope also changes very little with changes in discharge, if measured over a reach of several miles. Thus, the Thackston-Krenkel equation is more adaptable for use in the prediction of changes in k_2 with changes in discharge. All that is required is an approximate relationship between average depth and discharge.

Limitations and Uncertainties

All of the equations have their limitations. The Thackston-Krenkel equation has questionable accuracy at low Froude numbers, in streams with slow velocites and deep depths, because there is no reliable data in this range to fit the equation to. The Churchill equation is definitely inaccurate in slow, deep streams, and it and the Dobbins and O'Connor-Dobbins equations are inaccurate in very shallow, swift streams.

The Thackston-Krenkel equation cannot be used at all in very slow, deep streams such as estuaries or reservoir backwaters, because the slope is so flat that it cannot be determined accurately. In these situations, the O'Connor-Dobbins equation seems to fit the data best, but is is uncertain whether or not this is good, because the data is of questionable accuracy.

Very low values of k_2 predicted by any of the equations are likely to be too low, because of the effect of wind. The wind will increase the true surface area and will cause surface currents, increasing the vertical diffusion rate. Thus, even perfectly still water can be reaerated by wind. However, the contribution of wind to the reaeration process is variable and unreliable, and cannot be depended upon to be present at all times.

All of the equations apply to relatively clean and unpolluted water, and all predicted values of k_2 should be reduced somewhat if the subject reach is highly polluted. The amount of reduction required may vary from 10% to 50%, but, in most cases, will probably be in the range of 20% to 30%.

All the equations apply to fully developed turbulent shear flow with a regular vertical velocity profile and do not apply to laminar flow. They also do not apply to conditions so turbulent that waterfalls, riffles, or "white water" is present, and bubbles are entrained directly in the water.

It is hoped that the current work at Georgia Tech on the measurement of reaeration coefficients will supply data which can be used as standards against which the present prediction equations can be compared and refined, if necessary, to improve their accuracy and reliability. The tracer method is certainly the most accurate method of measurement available, and should produce much reliable data to supplement the limited data presently available.

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Pollutant Effects on Reaeration

L. A. Neal

Introduction

Certain pollutants can alter the reaeration capacity of a stream.

This paper describes some of the results of laboratory studies dealing with the effect of pollutants on reaeration. The purpose of the research has been to test the sensitivity of physical gas transfer to pollution.

Basic Considerations

Aeration of water is a gas-liquid mass transfer process that takes place as the result of the combined effects of molecular diffusion of oxygen and physical mixing of the water.

The basic mathematical expression describing aeration of clean water is

$$\frac{dD_t}{dt} = -K_2 D_t \qquad (1)$$

in which D_t is the dissolved oxygen concentration deficit below the saturation limit, in mg/l, at time t, and K_2 is the gas transfer rate coefficient for oxygen in clean water (1/time). Equation (1) may also be written as

$$\frac{d(C_s - C_t)}{dt} = -K_2(C_s - C_t) \qquad (2)$$

in which C_s is the dissolved oxygen saturation limit for clean water, and C_t is the dissolved oxygen concentration at time t, both in mg/l. A derivation of equation (1), from basic considerations of the kinetics of gases, has been presented earlier (1).

Equation (2) is a formulation of oxygen transfer in a clean water system. Other complicating factors must be accounted for if one is to consider a polluted water. A general expression may be written as

$$\frac{d(\beta C_s - C_t)}{dt} = -\alpha K_2(\beta C_s - C_t) + r \qquad (3)$$

in which β is the ratio of dissolved oxygen saturation for the polluted water to that for clean water, α is the ratio of the gas transfer rate coefficient for the polluted water to that for clean water, r is the rate of dissolved oxygen utilization, with other terms as previously defined.

The beta (β) factor accounts for any difference between the actual and "book value" dissolved oxygen saturation limit. Standard Methods (2) shows the depression of oxygen solubility due to chloride ion concentration. Generally, however, the β factor is not readily predicted and must be determined experimentally.

The alpha (α) factor accounts for the fact that various pollutants can alter the ability of gas molecules to enter and escape water. This alteration causes the value of K_2 to vary under the same conditions of turbulence depending upon whether clean water or polluted water is being aerated. It has been pointed out (3) that the α factor is not only related to the pollutant constituents and concentrations but also the turbulent mixing regime within the fluid. This means that α determinations must either be conducted in the full-scale system or in a system that duplicates the turbulent mixing within the real system.

With appropriate modification, the foregoing expressions also describe the transfer of other gases. Specifically, consider a dissolved tracer gas, krypton-85, which has been added to the water. The amount of kyrpton-85 present in the atmosphere above the water can be taken to be zero, for practical purposes. Hence, the driving force for gas transfer will be just the partial pressure of the dissolved krypton-85 in the water. Thus, in the case of desorption of the tracer gas we can write

$$C_t = C_o e^{-Kt} \qquad (4)$$

where C_t is the concentration of the dissolved tracer gas remaining in the water at time, t, C_o is the concentration at $t = 0$, and K is the gas transfer rate coefficient for the tracer gas in clean water.

Equation (4) can be modified to describe the desorption of the tracer gas in polluted water so that

$$C = C_o e^{-\alpha Kt} \qquad (5)$$

where alpha (α) is the gas transfer rate coefficient for the polluted water to that for clean water.

It has been shown (4), both experimentally and theoretically, that for the same conditions of turbulence

$$\frac{\alpha K}{\alpha K_2} = 0.83 \pm 0.04 \qquad (6)$$

and this is the basis for using krypton-85 as a tracer gas for oxygen in aeration studies. The numerical constant, 0.83, in equation (6) is independent of the degree of turbulent mixing, independent of the directions in which the two gases (krypton-85 and oxygen) happen to be moving and independent of temperature within the range 10 to 30°C. Additionally, the ratio has been verified (5) in a biologically active wastewater.

Since the pollutant effect on krypton-85 transfer can be equated to the pollutant effect on oxygen transfer, the tracer gas is well suited to the measurement of alpha factors in an experimental system.

Measurement of Alpha Factors

In order to measure the effect of pollutants on the gas transfer rate coefficient, an open top mechanically mixed reactor was constructed as shown in Figure 1. The reactor was immersed in a constant temperature (20°C) bath. The reactor was a four liter reaction kettle, and was operated with a water volume of 3600 ml.

One pollutant effect test consisted of two reactor runs. The first run was conducted on clean water and a krypton-85 transfer rate determined. The second run was conducted on the polluted sample under conditions identical to those of the clean water run. For each pollutant effect test, the alpha factor was computed from

$$\frac{K_{polluted}}{K_{water}} = \alpha \qquad (7)$$

where $K_{polluted}$ was the observed krypton-85 transfer rate coefficient for the polluted sample and K_{water} was the observed krypton-85 transfer rate coefficient for the clean water sample.

Operation of the Pollutant Effects Reactor

In the typical experiment, the reactor system was initially dismantled, cleaned, rinsed thoroughly with distilled water and reassembled. The reactor was then filled with 3600 ml of distilled water and allowed to stir. When thermal equilibrium was achieved, the distilled water was dosed with a homogeneous mixture of dissolved krypton-85 gas and tritiated water molecules with both tracers contained in about two ml of distilled water. The tritiated water was used to account for dispersion, as described elsewhere (1). After allowing the tracer dose a few minutes to disperse, the first sample was taken directly from the reactor with a 2-ml pipette immersed so that it filled by gravity to

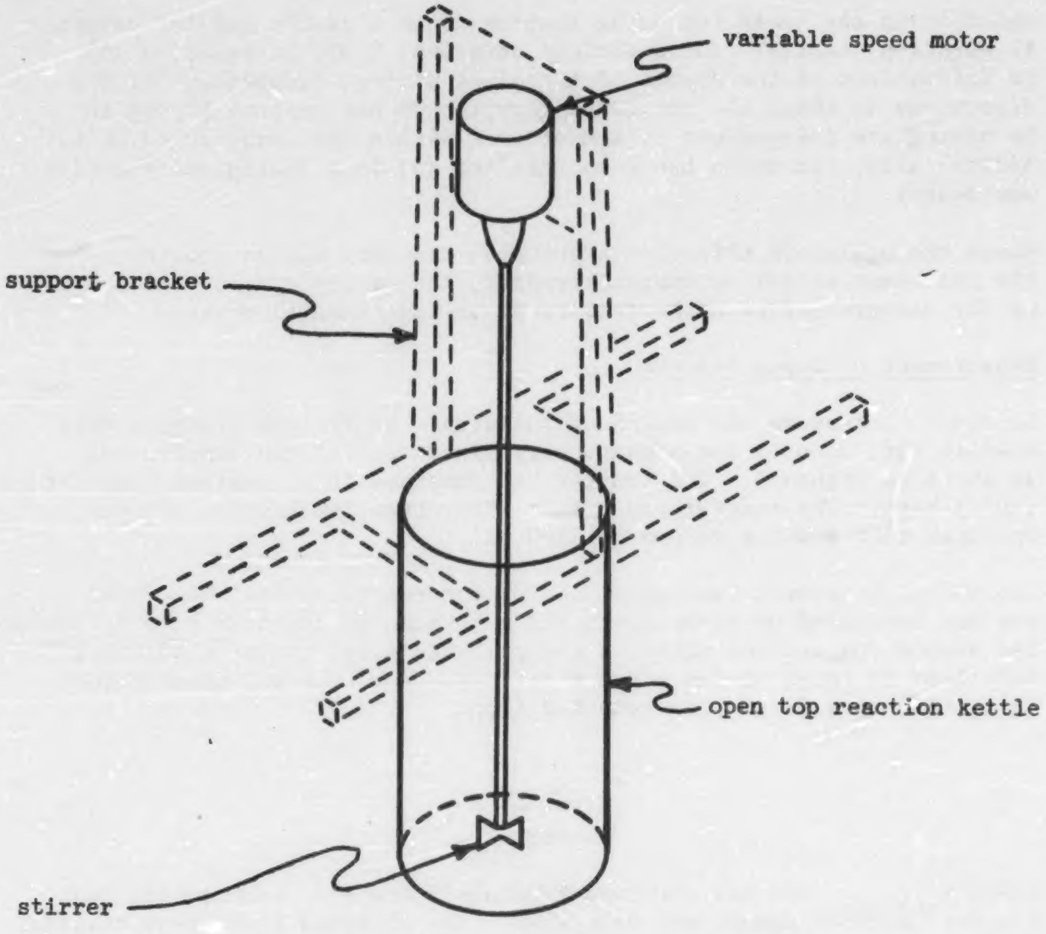


FIGURE 1
REACTOR ARRANGEMENT
(pollutant studies)

a point above the fiducial mark. The pipette was then removed from the reactor and excess sample wasted until the liquid reached the mark. The 2-ml sample was then transferred to a 25-ml counting vial that had been previously filled with 10 to 15 ml of liquid scintillation fluid. The transfer technique was designed to minimize the loss of dissolved kyrpton-85 gas. After the transfer was complete, the vial was then filled with the scintillation fluid, capped, and loaded into a liquid scintillation counter for measurement of the tritium and krypton-85 concentrations.

Subsequent samples were taken in the same manner until the run was complete.

The reactor was then drained, refilled with the polluted sample and the second run conducted just as the first described above.

Each tracer sample was subsequently counted for 10 minutes at least two different times and the concentration ratio of krypton-85 to tritium obtained.

Calculation of Krypton-85 Transfer Rate Coefficients

For each run, the krypton-85 to tritium concentration ratios were plotted as a logarithmic function of time. The line of "best fit" was then obtained by the method of least squares. The slope of this line was reported as the krypton-85 transfer rate coefficient for the particular run. The alpha factor was then calculated from equation (7).

A typical test result for the pollutant linear alkylate sulfonate (LAS) is shown in Figure 2.

Pollutant Studies

The detergent surface active agent LAS was selected for a series of pollutant effect studies in the reactor system previously described. A total of 12 tests (24 runs) have been conducted with LAS concentrations up to 11 mg/l. All 24 runs were conducted at one mixing speed so that the effect of LAS concentration on gas transfer could be studied.

In order to determine the relative magnitude of pollutant effects on the reaeration of natural river waters, 10 reactor tests (20 runs) were conducted on highly polluted samples from the South and Chattahoochee Rivers in the vicinity of Atlanta. The tests were conducted at several different mixing speeds so that the range of alpha values in the rivers could be estimated.

Experimental Results

Although analysis and interpretation of the data obtained is not complete, the results thus far provide considerable insight regarding the effect of pollutants on the reaeration of water.

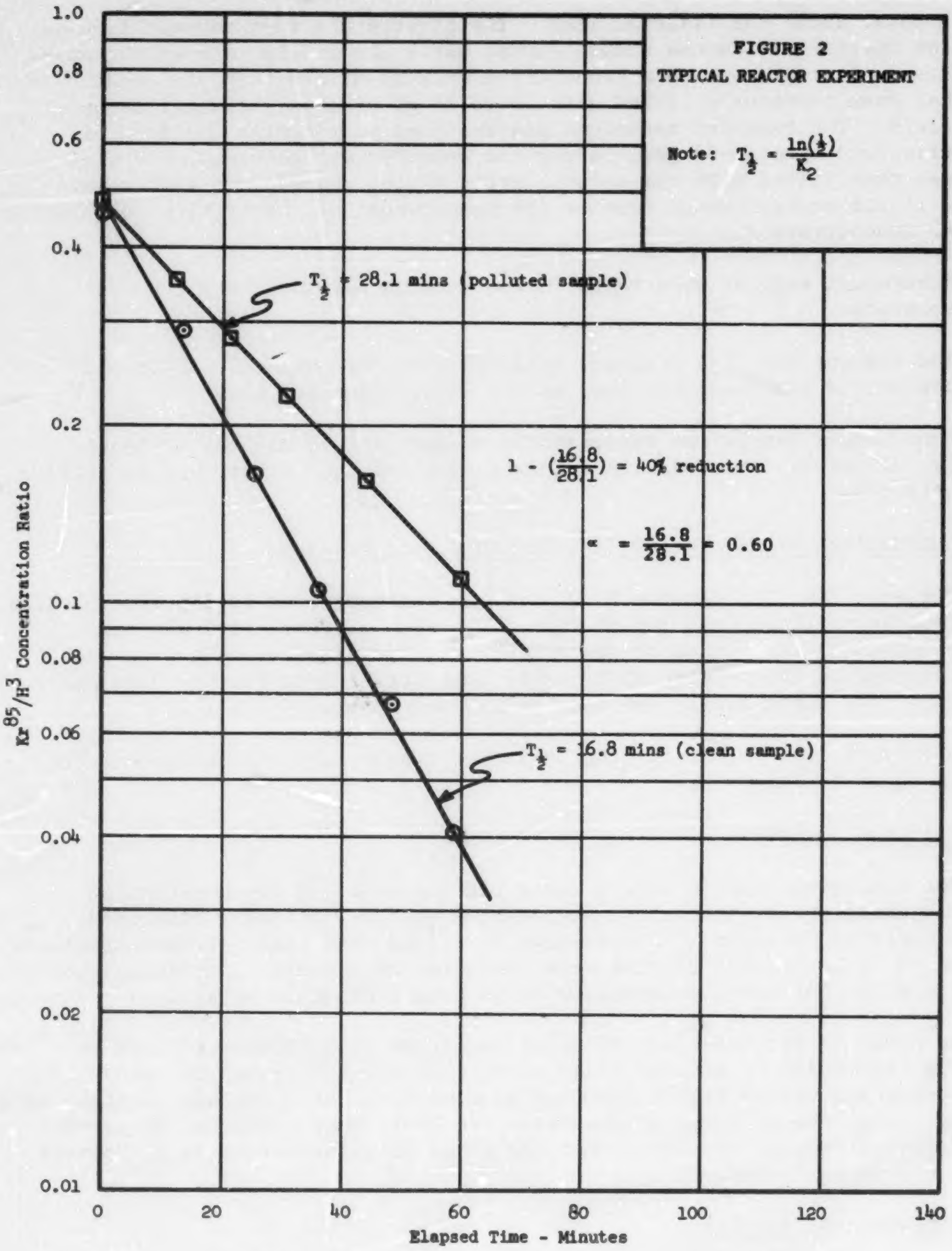


FIGURE 2

TYPICAL REACTOR EXPERIMENT

It is obvious from the summarized LAS data in Table 1 that the alpha factor decreases as LAS concentration increases. Figure 3 is a graph of the 12 LAS tests and appears to follow a smooth relationship as shown.

A few tests (not reported) have been conducted on water containing 10 mg/l LAS with a different mixing speed for each test. The results of these tests indicate that an increase in mixing speed tends to decrease the alpha factor for the same LAS concentration.

Table 2 is a summary of the test results obtained for 10 alpha determinations on South and Chattahoochee River water samples. The station identifications used in Table 2 correspond to those used in Tsivoglou's field investigations of reaeration capacity.

In February of 1970, samples were collected from four stations on the South River for the purpose of determining the LAS concentration at each point. The sampling stations corresponded to those used in Tisvoglou's field investigations and the results of the LAS determinations are in Table 3.

Discussion and Summary

It is clear from test results that the reaeration rate in a stream can be significantly reduced by the surface active agent, LAS.

The reaeration rates in the highly polluted reaches of the South and Chattahoochee Rivers are lower than they would be in the absence of pollution.

For a particular stream section, it would seem that the measured reaeration rate would vary from one study to another if the pollutant effect (alpha) was not the same for each study. Apparently, at least one of the tracer studies conducted in the South River was influenced by such a variation in the alpha factor. Figure 4 is a comparison of two different tracer studies on the same stretch of the South River. Measured flows and travel times were identical, for practical purposes, during both studies. From Figure 4 it is obvious that a discrepancy existed between the observed reaeration rates for the two studies. The difference in reaeration capacity was greater in the upper reaches than in the lower reaches. The observed reaeration rate for the last reach (E to F) is essentially the same for both studies. It is important to note that the City of Atlanta South River Sewage Treatment Plant discharges directly to the South River just above Station A (see Figure 4). Apparently, some pollutant released from the South River Plant (during Study X) caused the reaeration capacity to be lower for Study X than for Study VI. The pollutant effect variation diminished in the lower reaches (see Figure 4), due to dilution and possible degradation of the pollutant in the stream.

The LAS data in Figure 3 are considered as typical and indicate that

Table 1. Summary of Alpha Tests on Linear Alkylate Sulfonate in Distilled Water

Test	Run	LAS Conc. (mg/l)	K (1/hr) @20°C		$\alpha = \frac{K_{water + LAS}}{K_{water}}$
			water	water + LAS	
8	A	0.0	1.791		0.80
8	B	3.0		1.431	
9	A	0.0	1.946		0.74
9	B	3.8		1.437	
10	A	0.0	1.805		0.76
10	B	4.8		1.373	
11	A	0.0	1.904		0.68
11	B	6.0		1.289	
12	A	0.0	1.900		0.65
12	B	11.0		1.240	
13	A	0.0	2.065		0.92
13	B	0.96		1.908	
14	A	0.0	2.125		0.86
14	B	2.0		1.832	
15	A	0.0	2.034		0.78
15	B	5.2		1.584	
16	A	0.0	2.127		0.73
16	B	7.0		1.561	
17	A	0.0	1.903		0.75
17	B	9.2		1.426	
18	A	0.0	2.016		0.70
18	B	9.5		1.408	
19	A	0.0	2.04		0.69
19	B	8.3		1.40	

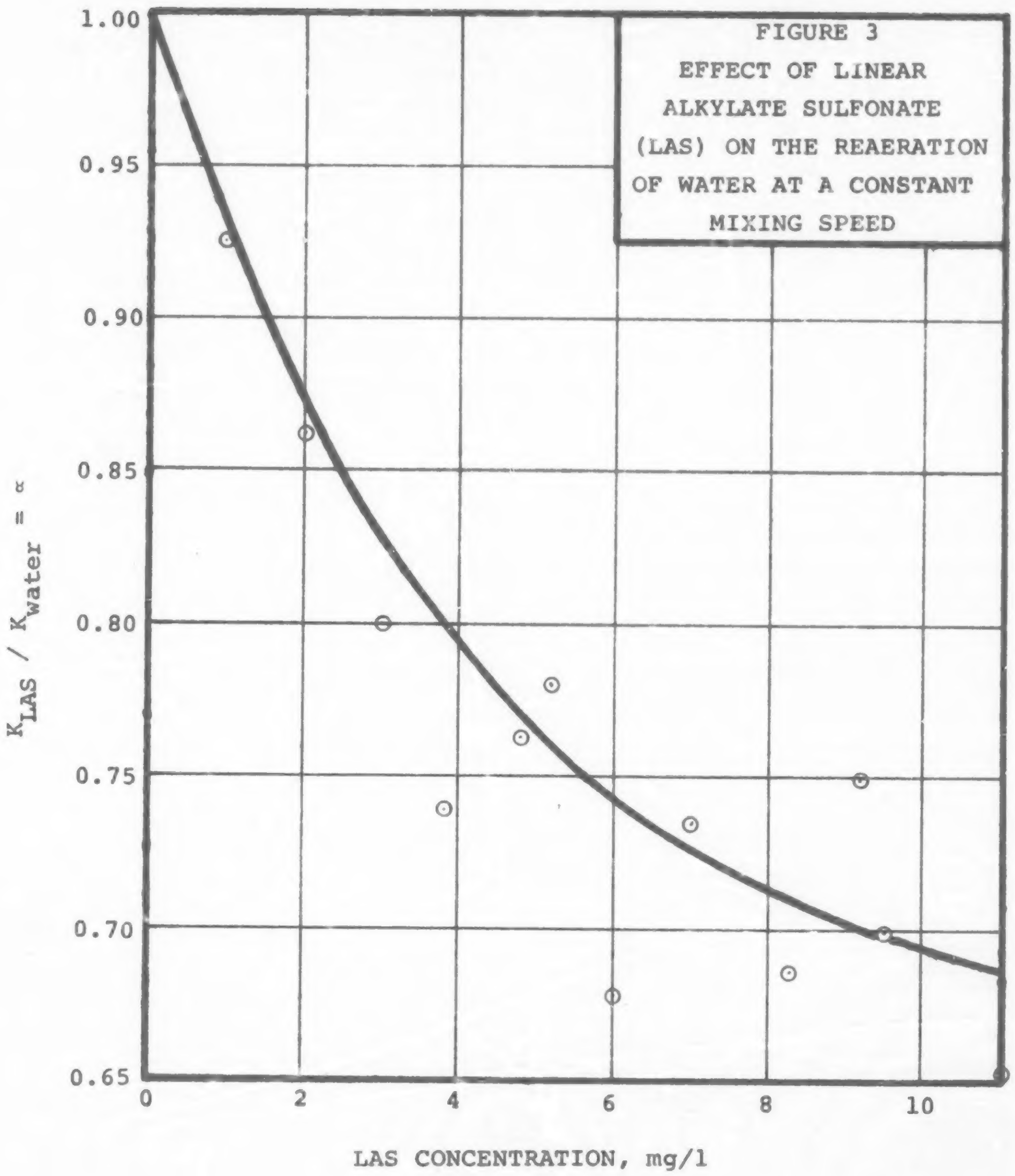


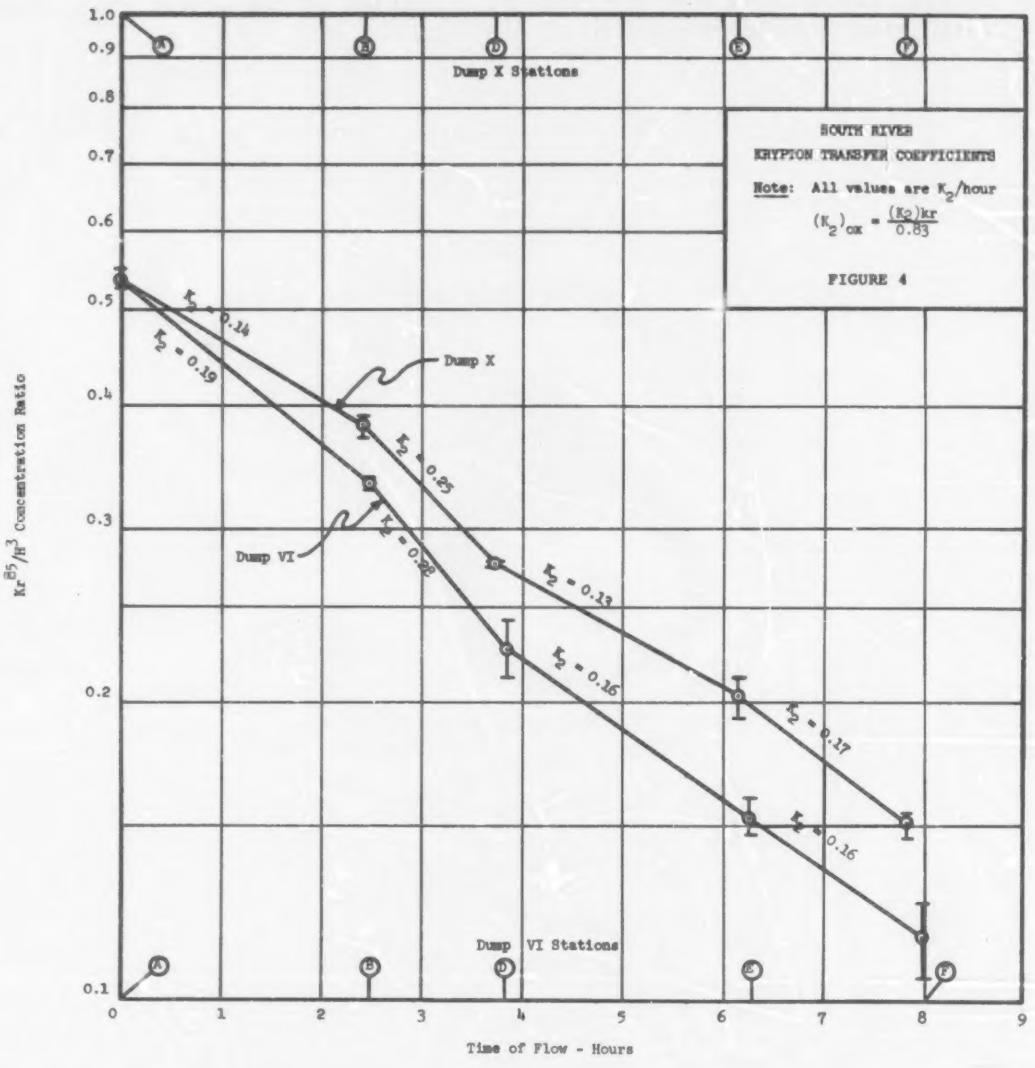
FIGURE 3
EFFECT OF LINEAR
ALKYLATE SULFONATE
(LAS) ON THE REAERATION
OF WATER AT A CONSTANT
MIXING SPEED

Table 2. Summary of Alpha Tests on Chattahoochee and South River Water Samples

River	Station	K (1/hr) @20°C "clean" water	K (1/hr) @20°C river water	$\alpha = \frac{K_{river}}{K_{clean}}$
South	G	0.937	0.796	0.85
Chatt.	O	2.29	1.48	0.65
Chatt.	O	2.47	1.48	0.60
Chatt.	O	0.52	0.40	0.77
Chatt.	O	1.92	1.17	0.61
Chatt.	O	0.49	0.43	0.88
South	between A & D	0.83	0.68	0.82
South	J	1.04	0.74	0.71
Chatt.	above O	0.83	0.69	0.83
Chatt.	O	2.04	1.63	0.80

Table 3. Summary of Linear Alkylate Sulfonate Data, South River

Station	LAS Conc. (mg/1)
G	2.6
below H	2.7
J	2.0
L	1.8



SOUTH RIVER

KRYPTON TRANSFER COEFFICIENTS

Note: All values are K_2/hour

$$(K_2)_{ox} = \frac{(K_2)kr}{0.83}$$

FIGURE 4

the observed reaeration rates in the South River studies of Tsivoglou were probably lower than the rates that would have been observed for clean water conditions.

Current research effort involves investigation of the effect that other pollutants have on reaeration.

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Observed vs. Calculated Reaeration Capacities of Several Streams

J. R. Wallace

We have prepared comparisons between our measured K_2 values and K_2 values computed from several of the formulas that are available. Specifically we have made and will show comparisons with three equations. One of these we refer to as the O'Conner-Dobbins equation, which was published in 1956. It is like neither the O'Connor nor the Dobbins equation that Dr. Thackston presented earlier, but it is similar to the Churchill equation in that it has a velocity in the numerator and a depth in the denominator. The velocity is raised to the 1/2 power and the depth to the 1 and 1/2 power. The other two equations we will be looking at in terms of their predictions are Churchill's equation, as shown on Table I, and then Dr. Thackston's equation which is also shown on Table I.

Now I would like to make just a couple of remarks about how we determine the values of the parameters in these equations. In the equations that we are considering we have three different parameters, two in two of the equations and three in the third. We have velocity, depth, and slope. The velocity used in these equations was simply the length of the reach divided by the time of passage as measured from our dye studies. The depth of flow was the average of the average depth at each cross section. For example, maybe we had a reach that was, let's say, 5,000 ft. long; within that we would have had 10 - 500 ft. stations at which we measured the hydraulic properties. At each of these 500 ft. stations we determined an average depth by dividing the cross sectional area by the width of the stream. The value of the depth which we subsequently used in the calculation was the average of these average cross sections depths. We determined slope as the difference in elevation at the upstream and downstream end of the reach divided by the length.

In our discussions our results should be separated into two categories depending upon the hydraulic characteristics that were found in each of the reaches. The first category would be that of a reach that has a relatively uniform cross section with unbroken surface. The second category would contain those reaches that are highly variable in cross section and are reaches which contain features that create high gas losses, i.e., reaeration rates, such as rapids, shoals, and falls. I would like to point out that it is not intuitively clear in every case where we make the break in category. That is, where does a stream become variable or turbulent to the degree that we take it out of one classification and put it into another. If you will keep that limitation in mind as we go along

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TABLE I

Comparison of Observed and Predicted Results

		Reaeration Rate Coefficient, K/hr*			
River	Reach	Observed	O'Connor	Churchill	Thackston
Flint	01	0.37	0.12	0.06	0.29
Flint	12	0.52	0.18	0.13	0.24
Flint	24	0.27	0.11	0.08	0.15
Flint	46	0.10	0.08	0.05	0.09
South	AE	0.22	0.18	0.18	0.13
South	EG	0.15	0.14	0.13	0.10
South	GJ	0.30	0.08	0.06	0.11
South	JM	0.17	0.09	0.07	0.10
Patuxent	14	0.13	0.14	0.08	0.12
Patuxent	47	0.13	0.11	0.06	0.12
South	GH	0.64	0.12	0.12	0.17
South	GT_2	1.6	(0.25)	(0.27)	0.36
South	KL	2.5	(0.10)	(0.07)	0.47
Flint	R_1R_3	2.0	(0.43)	(0.35)	0.64
Flint	1P1	12.7	--	--	2.4
Flint	2P2	2.7	--	--	0.38
Chattahoochee	OC	0.031	0.037	0.035	0.022

* 25°C

Churchill et al.: $K_L = 0.543 \dfrac{V^{0.969}}{H^{1.673}}$

O'Conner-Dobbins: $K_2 = 0.573 \dfrac{V^{0.5}}{H^{1.4}}$

Thackston: $K_2 = 1.122 \left\{ 1 + \left[\dfrac{V}{\sqrt{gh}} \right]^{0.5} \right\} \dfrac{Sg}{H}$

it might be helpful. First of all I think we can conclude that the mathematical models that we have examined provide predictions that are closer to the observed K_2 values in the uniform reaches, and this is certainly what I think would be expected. The equations, to some extent, have been based upon assumptions about the uniformity of the channel and on the assumption of unbroken surfaces. One author, Churchill, points out that his equation should not be used in any reach in which there is white water rapids. I am just going to give a summary of the data here and then we'll look at a graphical comparison. If we start up at the first line of Table I we see that for the Flint River, reach 0 to 1, the observed value of K_2 was 0.37, and the O'Conner equation predicts 0.12, Churchill 0.06 and Dr. Thackston's equation 0.29. Likewise, on the second reach of the Flint River, which is similar in nature hydraulically, we observed a value of 0.52, O'Conner's equation predicted 0.18, Churchill 0.13 and Thackston 0.24.

The hydraulic characteristics of these reaches are such that they do contain some fast moving water and the depth is relatively small, typically on the order of a foot or less. However these sections do not contain the waterfalls that we talked about before. Those are included in a later entry. As we go on down the list we see on the Flint River an observed value of 0.27, O'Conner 0.11, Churchill 0.08 and Thackston 0.15. Again on the Flint, this is down on the lower reaches where we're getting into slower moving water and somewhat deeper channel, 0.1 observed, 0.08 O'Conner, 0.05 Churchill, Thackston 0.09. Then we move over to the South River, which is a little larger stream, where the velocities are somewhat similar to these on the Flint. We have here 0.22 observed, 0.18, 0.18 and 0.13 predicted. I will have to ask Dr. Thackston later why his value at this point seems to be lower where it was higher in the others, but maybe we can get into this at a later time. Moving on down the South, we have 0.15 observed, 0.14, 0.13 and 0.10 predicted. 0.30 observed, 0.08, 0.06, and 0.11 predicted, and 0.17 observed, 0.09, 0.07, 0.10 predicted, and similarly until we get on down to section GH. At this point we start picking up more rapids. The K_2 observed goes up and the equations at this point are significantly under predicting the observed K_2. On the South River K_2 now goes up to 0.16, which is quite high, and at this point I place parenthesis around the Churchill and O'Conner results because these are in quite turbulent waters, and as I have pointed out, Churchill stated that his equations should not be used under these conditions. The only reason for not also setting apart Dr. Thackston's predictions for this stretch would be that they give much higher values than the others, and we thought there was something fundamentally different in his equation in comparison with the others. As we go on down the table to the Flint River, we see values that include reaches with waterfalls and we don't even make a comparison with the other two equations. We have put in the values, for what they are worth, into Dr. Thackston's equations, and the equation predicts quite high values at this point, but certainly nothing that would come close to the actual measured values. We certainly don't criticize his equation, because we are using it in an instance here for which it was never intended to be used. We have only one result from the Chattahoochee shown in this table. The value for the Chattahoochee falls off quite significantly and to two decimal places then we would have 0.03. This then, is a summary of the results that we have.

To give a little better comparison between the equations, the predicted values, and our measured values I have eliminated all sections that contain any white water, and only the most uniform sections are included in this next comparison. On Figure I you see plotted the observed K_2 per hour and the predicted K_2 per hour, with the various equations indicated by different symbols. The tendency, as you can see, is for the predictive models to appear low by something on the order of 30 to 50 percent. These are probably too few data to draw any conclusions, but I would say that the error tends to decrease as we get into lower values of K_2. The other conclusion that I think we can draw from this is that, at least for these data, no one model appears to be better than the other in its predictive abilities over the range of data that are here presented. Those are the only comments that I had to make on our comparisons, if I can have the lights back on maybe we can have a few minutes to discuss these results. Are there any questions or comments from anyone?

Discussion

Question: Much of your data is from streams where the flow is not uniform; is that correct? (JRW) That's right, and that's why I think it is important not to draw any overriding conclusions about the applicability of the predictive model. We must bear in mind at this point what the limits of these data are. As we continue with our studies on the Chattahoochee, which is a larger, slower moving stream than the ones shown on the table, the equations may give better results. I think you are absolutely right in pointing out the limitations of the comparison.

Question: Are there not many predictive models that you have not included in your comparison? (JRW) We certainly haven't tried to include every formula we could find in this comparison. What we have done is use those that are most talked about and discussed. At least in my work in this area, these are the ones that have been most often brought up and so that's the reason we selected the ones that we did. A number of others that I knew of could have been examined. Maybe before we get finished with this project we can make an attempt to collect all of the formulations that we can find and look into the predictive power of each when all our data is worked up.

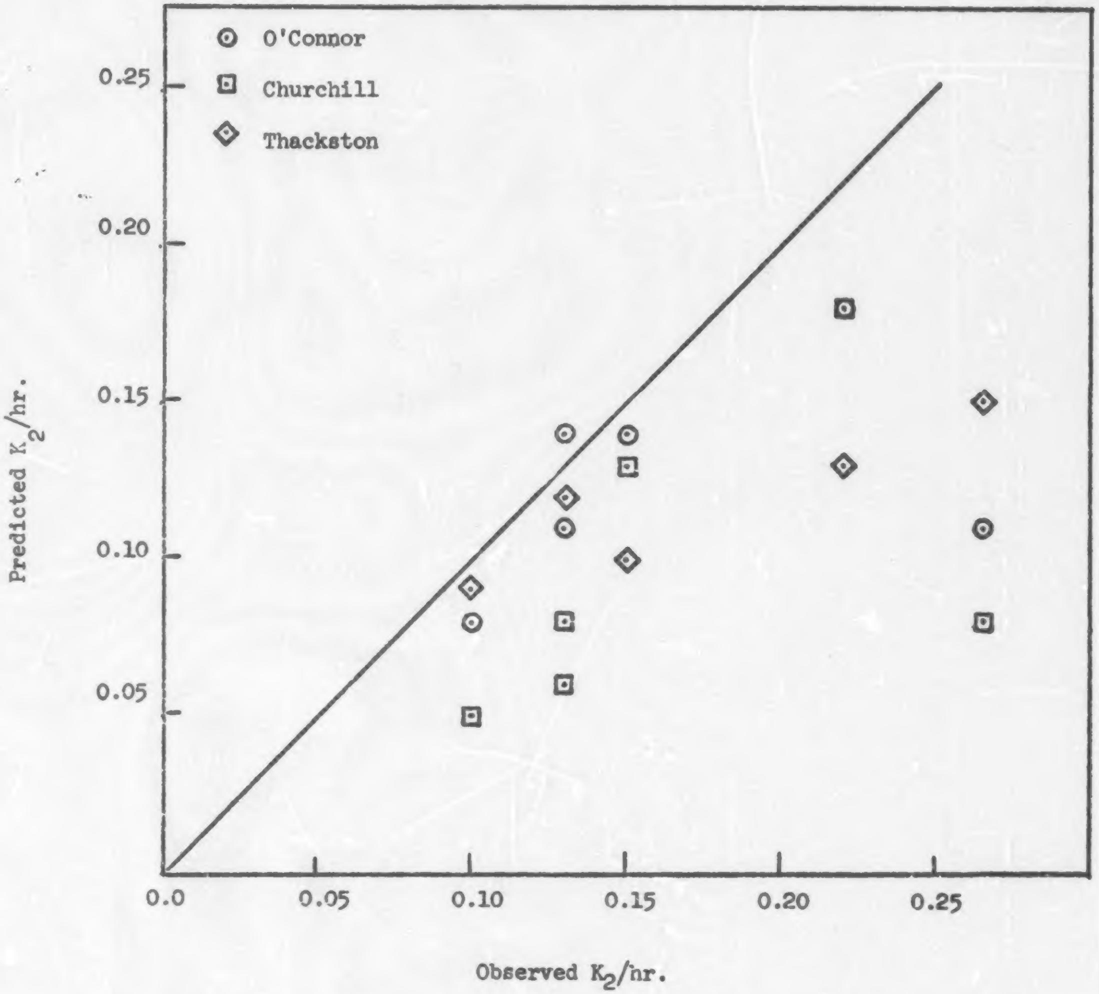

FIGURE I--OBSERVED VS CALCULATED REAERATION COEFFICIENTS

Relationships Between Hydraulic Properties and Reaeration

E. C. Tsivoglou

As indicated earlier, the main purpose of this research has been to evaluate the basic relationships between the reaeration capacity of a stream and its hydraulic properties, with the practical aim of developing the capability to predict reaeration on the basis of field measurement of appropriate hydraulic properties. We have seen that the rate of reaeration is directly proportional to the rate of water surface replacement. So in order to accomplish the above objective, we need to define the rate of water surface replacement in terms of those hydraulic properties that cause surface replacement, and then it is necessary to select or develop ways to measure hydraulic properties with accuracy in natural streams.

One of the real problems that we have encountered involves the basic meaning of certain traditional measures of hydraulic properties - the meaning specifically in terms of water surface replacement. Stated another way, sometimes the very method of measurement of a hydraulic property, or the method of computing it, modifies its real meaning in the physical sense. For example, as indicated earlier, if we consider a section of a natural stream, the way in which we measure the water depth affects its real meaning in terms of its relationship to the actual rate of water surface replacement. The measure, for instance,

$$\text{Depth} = \frac{\text{Occupied Channel Volume}}{\text{Surface Area}}$$

is valid only if there is complete and homogeneous mixing of all of the water in the channel, especially vertical mixing. To consider an extreme, in a stratified channel or reservoir the whole volume is separated into hydrodynamic regions, the depth that is effective in regard to surface replacement (and reaeration) is much smaller than the above expression would imply, and reaeration of the lower region is virtually nil. In many natural streams of relatively small slope the water depth that is effective in regard to surface replacement and reaeration is considerably smaller than the measurable whole depth of flow, due to poor vertical mixing.

Let us consider some of the hydraulic properties of natural watercourses, the possible ways of measuring them, and the resulting meanings or implications as regards reaeration capacity.

Velocity - Most of the available models for predicting reaeration capacity include the "mean velocity", either directly or indirectly, and at first glance it seems obvious that the rate of water surface replacement ought to be a function of the velocity. But on closer inspection

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two questions present themselves: first, how shall this "mean velocity" be measured, and secondly, does this method of observation affect the usefulness of the result as a measure of surface replacement? A third question, namely, is velocity a basic property that causes surface replacement, is perhaps the most important question of all.

There are at least two commonly used methods of obtaining the "mean velocity". The first involves direct physical measurement of the velocity at a number of locations in a stream cross-section by the use of a current meter; if enough such measurements are made, a reasonably accurate measure of the average forward velocity through that cross-section can be obtained. If, then, this procedure is repeated at a sufficient number of cross-sections in a specified length of stream channel, the results can be combined to obtain a reasonably accurate estimate of the mean forward velocity of flow that prevails throughout the length of the stream section. This procedure is subject to certain obvious sources of error relating especially to the statistical adequacy of the number of observations made in any one cross-section, the statistical adequacy of the number of cross-sections involved, and the accuracy of the current meter observations when forward velocities are relatively small. But in addition, and even more importantly, there would appear to be legitimate question as to whether the forward velocity is that velocity that is most nearly related to the rate of surface replacement.

The other commonly used method of obtaining the "mean velocity" involves measurements of the distance travelled and the time of flow. The distance travelled can be obtained readily and with quite adequate accuracy from USGS quadrangle sheets or by field survey; the time of flow can be measured with great accuracy by the use of dye tracers. The resulting "mean velocity", the distance divided by the time, is relatively precise because of the precision of the measures involved. Depending upon the degree of homogeneity and completeness of mixing in the channel, it is not necessarily the same "mean velocity" as that obtained by the first method outlined above, but it clearly reflects the actual forward velocity that is effective in the channel. Whether or not such a forward velocity adequately relates to the rate of surface water replacement is again open to serious question.

To summarize, although a measure of stream velocity can be obtained as outlined above, and although such a measure may be a "mean" in the usual sense, there is real question as to its usefulness as a representation of surface replacement. Perhaps a more meaningful measure for our purposes would be an estimate of the mean vertical velocity component in the stream channel, as this would seem to be more directly relatable to the rate of surface replacement. Again, however, the very method of observation could greatly affect the meaning and the usefulness of the result.

Depth - So far as the hydraulic properties are concerned, reaeration is a function only of the rate of surface replacement, and, hence, stream depth has importance only in terms of a possible relationship to the rate of surface replacement, and only then if mixing is complete and homogeneous. Although it seems unlikely, therefore, that depth itself is in any direct way a cause of surface replacement, let us examine the methods of observation. One has been outlined already, namely, the result of dividing the occupied channel volume by the whole surface area, and its meaning has been discussed. The other commonly considered method of observing the "mean depth" involves measurement of the dimensions of the stream cross-section by field survey, wherein the mean depth of the cross-section is obtained by dividing the observed cross-sectional area by the measured stream width. Of course, if an accurate measure of the "mean depth" of a length of stream channel is to be obtained by this means, a substantial number of cross-sections must be included for purposes of statistical adequacy. A substantial amount of field survey work is therefore involved. However, the result is subject to much the same criticism as was made for the volume/surface area method - the "mean depth" obtained is a measure of the effective depth only if there is complete and homogeneous mixing in the stream channel, and this requirement is not met in a large number of cases.

The available methods of obtaining an accurate measure of the effective mean depth of flow in a length of natural stream channel are tedious at best, even if there is complete and homogeneous mixing. In any event, the real meaning of such measures in terms of the rate of surface water replacement is not readily apparent, and it appears quite unlikely that depth itself has any causative relationship to surface replacement.

The velocity and depth of flow are the two hydraulic properties that appear directly in most of the available hydraulic models for predicting stream reaeration capacity. Other properties that appear either directly or by implication include the slope of the channel and channel bottom roughness.

Slope - The physical slope of a natural stream channel, namely the decrease in elevation per unit of channel length, is readily observable by field survey, although the fieldwork may be somewhat tedious and time-consuming. Surprisingly, although the slope would appear to be an important hydraulic feature, such measurements are not commonly made or available. Indeed, intuitively the slope of the stream channel would appear to be more nearly a determining or a causative property than most others - it is an independent property except where engineering works have modified it, and properties such as the velocity and depth of flow are functions of the slope rather than vice versa. In essence, the steeper the channel slope, the more violent the tumbling action that creates water surface replacement, and, hence, it appears that the channel slope should not only be related to the rate of surface replacement but should, in fact, be a basic cause of surface replacement. As indicated above, it can be measured with entirely satisfactory accuracy.

Roughness - One other property that would seem to be important in terms of water surface replacement is the physical channel roughness, in the sense that a very rough stream bed should create better vertical mixing than a smooth sandy stream bed. Of course, the bottom roughness cannot be measured directly or independently, and the available method of obtaining an estimate of bottom roughness, namely calculation by means of the Manning equation, is circuitous and subject to substantial error. In addition, the character of a stream bed, or its physical roughness, is not so independent a hydraulic property as might appear at first glance - in fact, the bottom character results from properties such as the velocity and the slope of the channel. Hence, although the bottom roughness may be related in some way to the degree of vertical mixing and the rate of surface replacement, it would not appear to be a basic property that independently causes surface replacement.

Certain of our experimental results have caused us to view the hydraulic properties in a somewhat different way that appears to have more promise in terms of developing a basic relationship between stream reaeration capacity and hydraulic properties. This point of view involves consideration of the relationship between surface replacement and energy dissipation.

Reaeration and Energy Dissipation

Consider a length of natural stream channel between two points, 1 and 2. The usual one-dimensional energy equation indicates that the amount of energy expended between the two points is

$$(E_1 - E_2) = \left(\frac{V_1^2}{2g} + z_1 + H_1\right) - \left(\frac{V_2^2}{2g} + z_2 + H_2\right) \qquad (1)$$

where V is the velocity in ft/sec, z is the elevation of the stream bed above mean sea level in ft, H is the depth of water in ft, and g is the gravitational constant in ft/sec^2.

Rearranging terms,

$$(E_1 - E_2) = \left(\frac{V_1^2 - V_2^2}{2g}\right) + \Delta h \qquad (2)$$

where

$$(z_1 + H_1) - (z_2 + H_2) = \Delta h \qquad (3)$$

and Δh is the change in water surface elevation between points 1 and 2.

With few exceptions, the difference in velocity head, $(V_1^2 - V_2^2)/2g$, is negligibly small compared to the change in elevation head, Δh. Hence, for most reaches of stream

$$(E_1 - E_2) \approx \Delta h \tag{4}$$

for practical purposes.

The rate of energy expenditure is just the amount of energy expended per unit time, or

$$\left(\frac{E_1 - E_2}{t_f}\right) = \left(\frac{\Delta h}{t_f}\right) \tag{5}$$

where t_f is the time of flow from 1 to 2.

It has also been shown in our earlier work that

$$K_2 = an\frac{A}{V} \tag{6}$$

where K_2 refers to the gas transfer coefficient for any gas, the constant, a, refers to the molecular properties of the gas and the quality of the water, n is the number of surfaces of area A replaced per unit time, and V is the whole volume of water. The product $(n\frac{A}{V})$ is therefore just the rate of surface replacement in cm^2 per second per cm^3 of volume, if metric units are employed.

It appears logical to suppose that the rate of water surface replacement will be related to the rate of energy dissipation, probably in a simple and direct way. We have therefore postulated as follows:

POSTULATE: The rate of water surface replacement is proportional to the rate of energy dissipation in open channel flow.

Using the expressions given in equations (5) and (6) above, the postulate may be expressed as follows:

$$\left(n\frac{A}{V}\right) = b\left(\frac{\Delta h}{t_f}\right) \tag{7}$$

where b is the necessary proportionality constant.

It now follows from equations (6) and (7) that

$$K_2 = c\left(\frac{\Delta h}{t_f}\right) \tag{8}$$

where c = ab.

Equation (8) is our basic model relating the reaeration coefficient, K_2, to the stream hydraulic properties. The coefficient K_2 actually refers to any gas, including krypton as well as oxygen, the only difference being the numerical magnitude of the constant, c. The hydraulic properties Δh and t_f can be measured directly and independently, as well as with quite satisfactory accuracy, for any length of stream channel. Hence, equation (8) and its underlying postulate given previously can be tested directly with field observations. A few of our relevant results will be presented for this purpose. However, before doing so, one other useful expression will be derived.

It has been noted earlier that for the length of stream between points 1 and 2

$$C_2 = C_1 e^{-K_2 t_f} \tag{9}$$

where C_1 and C_2 are the concentrations of dissolved gas at points 1 and 2. Replacing K_2 by its hydraulic equivalent from equation (8), we obtain

$$y = \frac{C_2}{C_1} = e^{-c\Delta h} \tag{10}$$

where y is just the decimal percent of dissolved gas remaining at point 2. It follows also that

$$(1 - y) = z = (1 - e^{-c\Delta h}) \tag{11}$$

where z is now the decimal percent of dissolved gas that has been lost between points 1 and 2. Equations (10) and (11) refer directly to the tracer gas, krypton, but may also be used to refer to the decimal fractions of the DO deficit remaining and satisfied, respectively.

Equation (11) is of very strong interest. It states, simply, that gas transfer in a turbulent natural stream is dependent only upon the change in water surface elevation, in the hydraulic sense. In other terms, at a given water temperature the amount of tracer gas that will be lost to the atmosphere in a specific length of stream channel, or the amount of DO deficit that will be satisfied, can be predicted on the basis solely of the change in water surface elevation. Alternatively, the numerical magnitude of K_2 can be predicted on the basis of the change in water surface elevation and the time of flow, according to equation (8).

Figure 1 is a graph of our presently available results from the tracer studies of the Flint, South and Patuxent Rivers, in which the mean value of K_{ox}(25°C) for the specific stream reach has been plotted as a function of the rate of energy dissipation measured as $(\Delta h/t_f)$. The reaches represented include waterfalls and rapids, as well as Panola Shoals and the relatively gentle mixing in reaches of the Patuxent. The range of values of K_{ox}, from 0.10 to 2.7 per hour, is quite large. As may be seen, these currently available results strongly support the straight line relationship predicted by equation (8), with a numerical value for the coefficient c of about 0.045 per ft. The single aberrant result (29, 2.7) is for the reach 2P-2 on the Flint River, which includes essentially the second waterfall, and this particular observation is regarded as presently still questionnable to the extent that the observed short time of flow may contain some error. This is being checked.

Figure 2 provides a separate test of the relationship predicted by equation (11), wherein gas loss was shown to be a function only of the change in water surface elevation. All of the individual tracer gas loss data obtained from the five separate dumps in the Patuxent River have been plotted, with the observed percent loss of tracer gas shown as a function of the change in water surface elevation between sampling points. The relationship predicted by equation (11) is clearly demonstrated by the data, as is the effectiveness of equation (11) for predicting gas transfer from energy dissipation. If these same results are plotted in the form required by equation (10) on semilog paper (log percent remaining vs elevation change), a straight line results, as predicted, and the degree of correlation is very good.

It is emphasized that the foregoing results must be regarded as preliminary at this time. Additional data are still being obtained, and final computation and correction of all of our observed results is still not complete. In brief, the research is still in progress, and these currently available data are presented here primarily to illustrate the direction that our research has taken and our current approaches. Several questions remain and are under investigation, relating largely to study of the observed spread of data about the predicted relationships. In particular, we know that the presence of pollutants does affect the numerical value of K_2. We have also observed that in at least one case different tracer dumps in the same river may produce excellent individual fits of the relationship predicted by equation (11) but still lead to different values of the coefficient c. Current effort in this research therefore involves investigation of the "finer structure" of the relationships that have been shown, and of the hydraulic properties and the water quality properties that may bring about such variability.

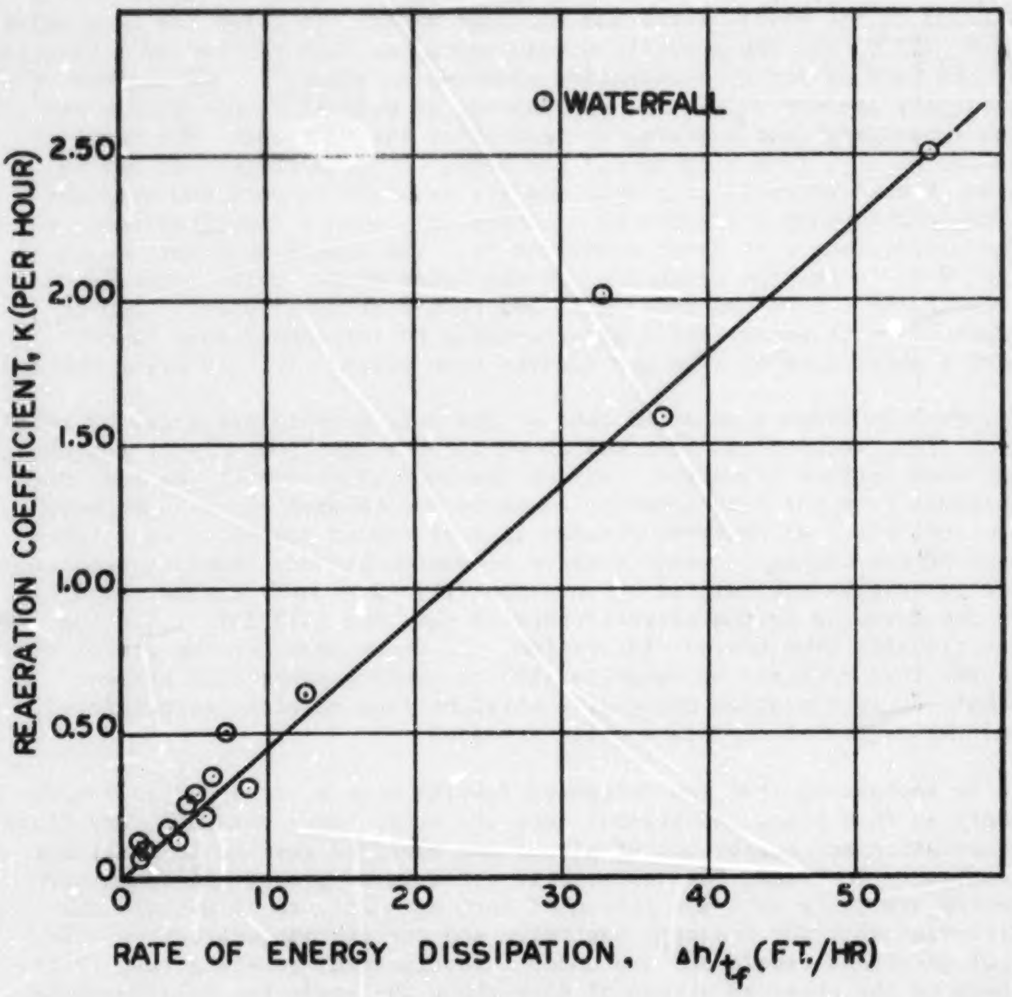

FIGURE 1

MEASURED REAERATION RATES
FLINT, SOUTH, AND PATUXENT RIVERS

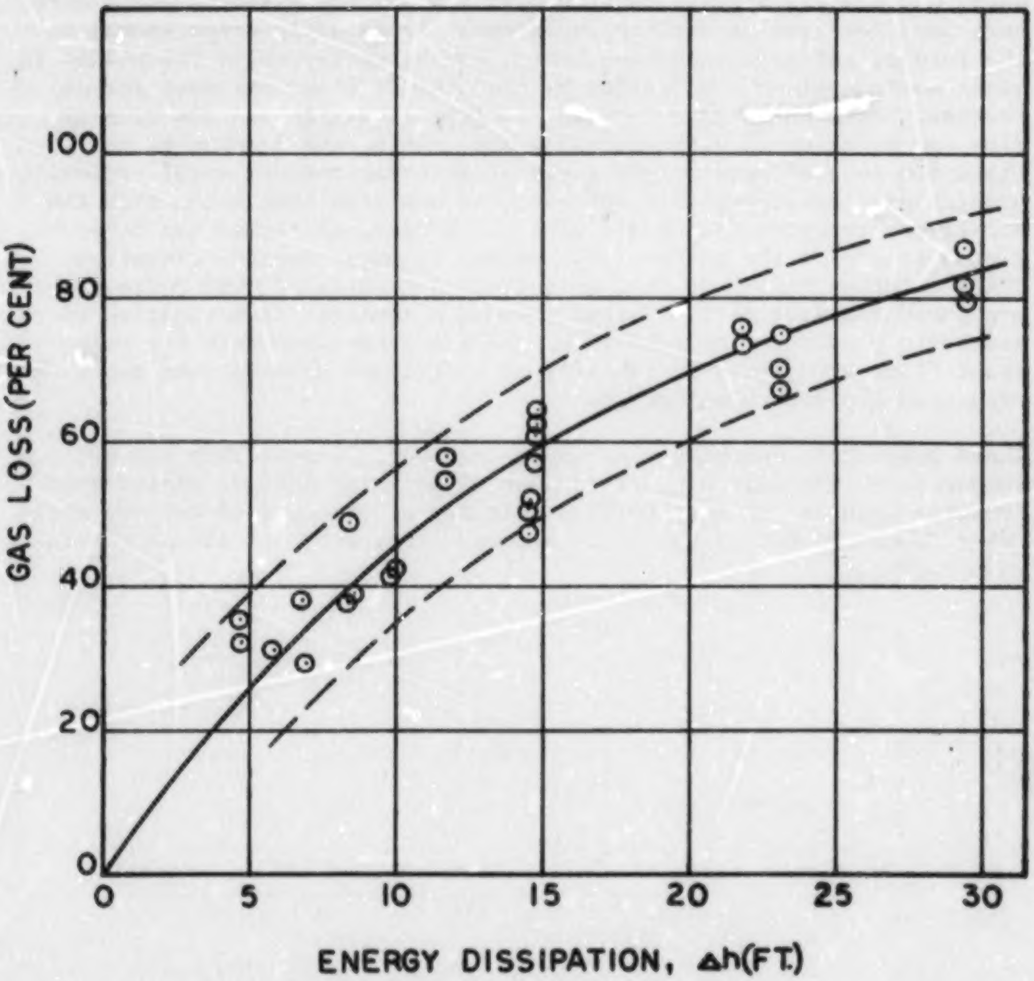

FIGURE 2

GAS TRANSFER FOR PATUXENT RIVER STUDIES

Summary

Stream reaeration has been shown to be directly related to the rate of water surface replacement in a natural stream. It has been postulated here that the rate of surface replacement is directly proportional to the rate of energy dissipation, which can be expressed as the change in water surface elevation divided by the time of flow, for most stream reaches. Both the change in water surface elevation and the time of flow can be observed with accuracy, and, hence, the reaeration coefficient can be predicted on the basis of observable fundamental hydraulic properties. Extending this approach, it has also been shown that the actual gas transfer that takes place in a reach of stream can be predicted solely on the basis of the change in water surface elevation. These relationships have been adequately demonstrated with currently available research data. Current research involves investigation of the hydraulic properties and the water quality properties that may bring about "fine structure" variability of individual observations about the predicted general relationships.

These considerations have also indicated that, because they are not fundamental hydraulic properties that cause water surface replacement, measures such as the mean forward velocity of stream flow and the mean water depth are not likely to be adequate indicators of stream reaeration capacity.